In compliance with the
Canadian Privacy Legislation
some supporting forms
may have been removed from
this dissertation.

While these forms may be included
in the document page count,
their removal does not represent
any loss of content from the dissertation.

University of Alberta

Greenhouse Gas Emissions from Commercial Composting

By

Mark Andrew Teshima

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of *Master of Science*

in

Bioresource and Food Engineering

Department of Agricultural, Food and Nutritional Science

Edmonton, Alberta

Fall, 2003

National Library of Canada
Acquisitions and Bibliographic Services
395 Wellington Street
Ottawa ON K1A 0N4
Canada

Bibliothèque nationale du Canada
Acquisisitons et services bibliographiques
395, rue Wellington
Ottawa ON K1A 0N4
Canada

Your file Votre référence
ISBN: 0-612-87895-3
Our file Notre référence
ISBN: 0-612-87895-3

The author has granted a non-exclusive licence allowing the National Library of Canada to reproduce, loan, distribute or sell copies of this thesis in microform, paper or electronic formats.

The author retains ownership of the copyright in this thesis. Neither the thesis nor substantial extracts from it may be printed or otherwise reproduced without the author's permission.

L'auteur a accordé une licence non exclusive permettant à la Bibliothèque nationale du Canada de reproduire, prêter, distribuer ou vendre des copies de cette thèse sous la forme de microfiche/film, de reproduction sur papier ou sur format électronique.

L'auteur conserve la propriété du droit d'auteur qui protège cette thèse. Ni la thèse ni des extraits substantiels de celle-ci ne doivent être imprimés ou aturement reproduits sans son autorisation.

Canadä

University of Alberta

Library Release Form

Name of Author: *Mark Andrew Teshima*

Title of Thesis: *Greenhouse Gas Emissions from Commercial Composting*

Degree: *Master of Science*

Year this Degree Granted: *2003*

Permission is hereby granted to the University of Alberta Library to reproduce single copies of this thesis and to lend or sell such copies for private, scholarly or scientific research purposes only.

The author reserves all other publication and other rights in association with the copyright in the thesis, and except as herein before provided, neither the thesis nor any substantial portion thereof may be printed or otherwise reproduced in any material form whatever without the author's prior written permission.

Dated: SEPTEMBER 24 *2003*

University of Alberta

Faculty of Graduate Studies and Research

The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for acceptance, a thesis entitled *Greenhouse Gas Emissions from Commercial Composting* submitted by *Mark Andrew Teshima* in partial fulfillment of the requirements for the degree of *Master of Science* in *Bioresource and Food Engineering*.

Dated: SEPTEMBER 29 *2003*

ABSTRACT

Emissions of the primary greenhouse gases (GHGs), carbon dioxide, methane, and nitrous oxide were measured from field-scale compost windrows using a static chamber method. After familiarization with the methodology, a second trial examined how GHG emissions could be influenced by varying management strategies, specifically turning frequency and C:N ratio. Laboratory-scale trials were conducted to evaluate their ability to model field-scale emissions. Finally, the suitability of the static chamber method for GHG measurement from composting was assessed.

The static chamber method was deemed a practical method of determining GHGs from composting, as values obtained agreed with previously published research. No significant differences were found between treatments in the second field trial. Cumulative emissions from field and laboratory trials were similar for carbon dioxide and nitrous oxide, but emission trends varied. The influence of environmental factors on the static chambers may make this collection technique more suited to controlled laboratory conditions.

"You see this goblet? For me, this glass is already broken. I enjoy it; I drink out of it. It holds my water admirably, sometimes even reflecting the sun in beautiful patterns. If I should tap it, it has a lovely ring to it. But when I put this glass on a shelf and the wind knocks it over or my elbow brushes it off the table and it falls to the ground and shatters, I say, 'Of course.' When I understand that this glass is already broken, every moment with it is precious.

-Achaan Chaa

"If you don't hate reality TV, you're not smart."

-Jim Rome

ACKNOWLEDGEMENTS

Many thanks to the following contributors to this thesis:

Dr. J.J. Leonard, for mentorship extending far beyond the science of composting, but also for invaluable lessons in professionalism and class, and for making my graduate studies experience memorable and enjoyable.

Dr. O.G. Clark, for his assistance in the refinement of several papers, posters and presentations, and for additional important contributions to my research.

Dr. J.J. Feddes and Dr. R.F. Grant, supervisory committee members, for their valuable input and contributions to this thesis.

Don Noot of Alberta Agriculture, Food and Rural Development, for volunteering his expertise to the project, and for allowing use of analytical equipment at the O.S. Longman Laboratory.

Dr. Salim Abboud, for allowing use of analytical equipment at the Alberta Research Council.

Jason Price, for showing me the ropes throughout grad school, and for partaking in countless hours of Whyte Ave. and conference "entertainment", and Joy Agnew, for her assistance with the "SD2K" pycnometer.

Max Amerongen, Brian Klein, Nash Goonewardena, Monica Molina for technical support.

Jody Forslund and Francine Hodder, for providing assistance throughout my time as a grad student, and for providing feedback on my awful jokes.

The Natural Sciences and Engineering Research Council of Canada, and KC Environmental, for funding this project.

Finally, thank you to my parents, Mutsuo and Patricia Teshima, and sister Kim Teshima, for their continued support, love, and encouragement.

TABLE OF CONTENTS

CHAPTER 1 INTRODUCTION ... 1
1.1 INTRODUCTION ... 1
1.2 LITERATURE REVIEW ... 3
1.2.1 Composting defined ... 3
1.2.2 The composting process ... 3
1.2.3 Benefits of composting and compost ... 4
1.2.4 Compost as a soil conditioner ... 5
1.2.5 Compost utilization in agriculture ... 7
1.2.6 Compost and biofiltration ... 7
1.2.7 Compost utilization and plant disease suppression ... 8
1.3 GREENHOUSE GAS AND GLOBAL WARMING ... 9
1.4 GREENHOUSE GAS AND COMPOSTING ... 13
1.4.1. Review of available literature ... 14
1.4.1. Synthesis ... 39
1.5 REFERENCES ... 39

CHAPTER 2 GREENHOUSE GAS EMISSION FROM COMMERCIAL COMPOSTING – FIELD SEASON 1: DECEMEBER 6, 2001 – MARCH 7, 2002 ... 43

2.1 INTRODUCTION ... 43
2.2 MATERIALS AND METHODS ... 44
2.2.1 Compost establishment ... 44
2.2.2 Compost off-gas sampling and GHG measurement using the static chamber method ... 46
2.3 RESULTS ... 49
2.3.1 Temperature monitoring ... 49
2.3.2 Field-scale GHG emissions ... 50
2.3.3 Lab-scale GHG emissions ... 55
2.4 DISCUSSION ... 59
2.4.1 GHG emissions from field-scale trial ... 59
2.4.2 GHG emissions from laboratory-scale trial ... 64
2.4.3 Laboratory-scale GHG emissions as a predictor of field-scale emissions ... 66
2.4.4 Evaluation of the static chamber method ... 67
2.5 CONCLUSION ... 69
2.6 REFERENCES ... 70

CHAPTER 3 GREENHOUSE GAS EMISSIONS FROM COMMMERCIAL COMPOSTING – FIELD SEASON 2 – JULY 3, 2002 – SEPTEMBER 26, 2002 ... 72

3.1 INTRODUCTION ... 72
3.2 MATERIALS AND METHODS ... 73
3.2.1 Compost establishment ... 73

3.2.2 Compost characterization 75
3.2.2.1 Determination of free air space and bulk density 75
3.2.2.2 Chemical analysis 75
3.2.3 Compost off-gas sampling and GHG measurement 76
3.2.4 Statistical analysis 78
3.3 RESULTS 78
3.3.1 Compost characteristics 78
3.3.1.1 Fee air space and bulk density profiles 78
3.3.1.2 Chemical analysis 80
3.3.2 Field- trial GHG emissions 81
3.3.3 Laboratory-scale GHG emissions 85
3.4 DISCUSSION 90
3.4.1 Physical properties 90
3.4.2 GHG emissions 91
3.4.3Variability in measurement of GHGs 96
3.5 CONCLUSIONS 96
3.6 REFERENCES 97

CHAPTER 4 EVALUATION OF THE STATIC CHAMBER METHOD FOR COLLECTION OF GREENHOUSE GASES FROM COMPOSTING 100

4.1 INTRODUCTION 100
4.2 MATERIALS AND METHOD 103
4.3 RESULTS 103
4.4 DISCUSSION 104
4.5 CONCLUSION 110
4.6 REFERENCES 112

CHAPTER 5 SYNTHESIS 113

5.1 SUMMARY 113
5.2 CONCLUSION 114
5.3 FUTURE DIRECTIONS 116
5.4 REFERENCES 116

LIST OF TABLES

Table 2.1 Compost feedstock physciochemical properties 45
Table 2.2: GHG emission data – field tial 50
Table 2.3: GHG emission data – lab trial 55
Table 3.1: Compost feedstock physicochemical properties and recipe volumes (low C:N treatment 74
Table 3.2: Compost feedstock physicochemical propoerties and recipe volumes (high C:N treatment) 74
Table 3.3: Summary of physicochemical methodologies 76
Table 3.4: Compost physicochemical analysis 80
Table 3.5: GHG emission data – field trial 81
Table 3.6: GHG emission data – lab trial 85

LIST OF FIGURES

Figure 2.1: CO_2-C emissions vs. composting days – field trial chambers 1-3 52
Figure 2.2: CO_2-C emissions vs. composting days – field trial chamber 4 and 5 52
Figure 2.3: CH_4-C emissions vs. composting days – field trial chambers 1-3 53
Figure 2.4: CH_4-C emissions vs. composting days – field trial chambers 4 and 5 53
Figure 2.5: N_2O-N emissions vs. composting days – field trial chambers 1-3 54
Figure 2.6: N_2O-N emissions vs. composting days – field trial chambers 4 and 5 54
Figure 2.7: CO_2-C emissions vs.composting days – laboratory trial indoor vessel 56
Figure 2.8: CO_2-C emissions vs. composting days – laboratory trial outdoor vessels 56
Figure 2.9: CH_4-C emissions vs. composting days – laboratory trial outdoor vessels 57
Figure 2.10: CH_4-C emissions vs. composting days – laboratory trial outdoor vessels 57
Figure 2.11: N_2O-N emissions vs. composting days – laboratory trial indoor vessels 58
Figure 2.12: N_2O-N emissions vs. composting days – laboratory trial outdoor vessels 58
Figure 3.1: Depth (m) vs. free air space (%) 79
Figure 3.2: Depth (m) vs. bulk density (kg m^{-3}) 79
Figure 3.3: FAS (%) vs. bulk density (kg m^{-3}) (High C:N/Turned Compost) 80
Figure 3.4: CO_2-C emissions vs. composting days – HT and T treatments (field trial) 82
Figure 3.5: CO_2-C emissions vs. composting days – T and C treatments (field trial) 82
Figure 3.6: CH_4-C emissions vs. composting days – HT and T treatments (field trial) 83
Figure 3.7: CH_4-C emissions vs. composting days – T and C treatments (field trial) 83
Figure 3.8: N_2O-N emissions vs. composting days – HT and T treatments (field trial) 84
Figure 3.9: N_2O-N emissions vs. composting days – T and C treatments (field trial) 84
Figure 3.10: CO_2-C emissions vs. composting days – indoor vessels (lab trial) 87
Figure 3.11: CO_2-C emissions vs. composting days – outdor vessels (lab trial) 87
Figure 3.12: CH_4-C emissions vs. composting days – indoor vessels (lab trial) 88
Figure 3.13: CH_4-C emissions vs. composting days – outdoor vessels (lab trial) 88
Figure 3.14: N_2O-N emissions vs. composting days – indoor vessels (lab trial) 89
Figure 3.15: N_2O-N emissions vs. composting days – outdoor vessels (lab trial) 89
Figure 4.1: CO_2 (ppm) vs. time (min.) – field trial 105
Figure 4.2: CO_2 (ppm) vs. time (min.) – laboratory trial 105
Figure 4.3: CH_4 (ppm) vs. time (min.) – field trial 106
Figure 4.4: CH_4 (ppm) vs. time (min.) – laboratory trial 106
Figure 4.5: N_2O (ppm) vs. time (min.) – field trial 107
Figure 4.6: N_2O (ppm) vs. time (min.) – laboratory trial 107

List of Symbols and Abbreviations

A	base area (m^2) (eqns. 3 to 6)
AAFC	Agriculture and Agri-Food Canada
BD	bulk density (kg m^{-3})
C	control treatment
ΔC	change in concentration (ppm) (eqns. 3, 6)
CCM	composted cattle manure
CG	Cleanit Greenit
CH4	methane
CFC	chlorofluorocarbon
C:N	carbon-to-nitrogen ratio
CO2	carbon dioxide
DOC	dissolved organic carbon (mg L^{-1})
DVB	divinylbenzene
EC	electrical conductivity (milliSiemens per centimeter)
ECD	electron capture detector
F	emission rate (g d^{-1}) (eqn. 5)
f	emission rate (g m^{-2} d^{-1}) (eqn. 4)
FAO	Food and Agriculture Organization of the United Nations
FAS	free air space (%)
FID	flame ionization detector
FTIR	Fourier Transform Infrared Spectroscopy
GC	gas chromatogram
GHG	greenhouse gas
g m^{-2} d^{-1}	grams per square meter per day (or per second, min, hour)
GWP	Global Warming Potential
H	high C:N treatment
HCFC	hydrochlorofluorocarbon
HFC	hydrofluorocarbon
Hg/Cd/Te	mercury/cadmium/telluride detector
HT	high C:N, turned treatment
IPCC	Intergovernmental Panel on Climate Change
kg m^{-3}	kilograms per cubic meter
M	molar mass (g mol^{-1})
m	air flow rate (m s^{-1}) (eqn. 5)
MC	wet basis moisture content
Mg ha^{-1} y^{-1}	megagrams per hectare per year
MSW	municipal solid waste
N	moles (Ideal Gas Equation)
$NaNO_2$	sodium nitrate
N_2O	nitrous oxide
NH_3	ammonia
NH_4^+	ammonium
NO	nitric oxide
NO_2^-	nitrite

NO_3^-	nitrate
P	pressure (atm) (Ideal gas equation)
PFC	perfluorocarbon
ppm	parts per million
PVC	polyvinyl chloride
R	Universal gas constant (0.08206 L atm mol^{-1} K^{-1}) (Ideal Gas Equation)
ρ	density (g mL^{-1}) (eqn. 3)
SF_6	sulfur hexafluoride
T	temperature (Kelvin)
T	turned treatment
T_A	ambient temperature during sample collection (C)
ΔT	change in time (min.) (eqns. 3,6)
TCD	thermal conductivity detector
TDLA	tunable diode laser analyzer
t ha^{-1}	tonnes per hectare
UNFCCC	United Nations Framework Convention of Climate Change
V	volume (L) (Ideal Gas Equation)
V	volume (m^3) (eqn. 6)
WSOC	water soluble organic carbon (ppm)
X_b	chamber concentration (g m^{-3})
X_e	ambient concentration (g m^{-3})

CHAPTER 1

Introduction

1.1 Introduction

As concerns escalate regarding anthropogenic greenhouse gas (GHG) production and global warming, a number of industries and sectors have been identified as significant GHG producers. In response, alternative technologies are being investigated to mitigate GHG production in these sectors. Waste management is one sector identified as a source of GHG production. Composting has frequently been put forth as a technology to reduce GHG emissions in this context. It is thought that well managed and designed composting facilities should not negatively affect air quality with respect to GHG emissions (AAFC, 1998). Until recently, however, scientific research regarding composting has been dedicated largely to improving compost quality and production. Process-oriented issues have not received much attention from the research community (Zeman, 2002), and very little objective data are available on topics such as the emission of GHGs from composting.

The purpose of the research described in this thesis was to collect additional baseline data on emissions from commercial composting of the three primary GHGs: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Data were collected during two field seasons; December 6, 2001 to February 28, 2002, and July 3, 2002 to September 26, 2002. In the first field season GHG emissions were measured from a single windrow at a commercial composting site: Cleanit Greenit Composting System Inc., a fully owned and operated subsidiary of KC Environmental Group Ltd., Edmonton, Alberta. During the

second field season, four differently managed research windrows were established to observe the effects of carbon-to-nitrogen (C:N) ratio and of turning frequency on the production of GHGs. The four treatments implemented in this trial were: 1) low C:N ratio, unturned (control); 2) high C:N ratio (44:1), unturned; 3) low C:N ratio (34:1), turned every two weeks; and 4) high C:N ratio, turned every two weeks.

Emissions from the compost were collected using the static chamber method, and analyzed for GHGs with gas chromatography (GC). Concurrent with both field seasons, laboratory-scale composting was conducted using the aforementioned treatments. Emissions of GHGs were measured at the laboratory scale for evaluation as a predictor of emissions from field-scale composting operations. During the second field season, bulk density (BD) and free air space (FAS) profiles were also established for the four research windrows using an air pycnometer (Agnew, 2002) that had been modified to simulate compressive loads, BD and FAS measurements were taken at simulated depths of 0, 0.5, 1.0, 1.5, 2.0, 2.5 and 3.0 m. Characterization of these properties allowed for an examination of how they may affect GHG emissions from composting.

Finally, the static chamber method was evaluated by analyzing repeated samples from the static chambers so as to determine an optimal sampling time at which flux was not inhibited nor emission rates underestimated.

1.2 Literature review

The following literature review begins with a concise definition of composting, followed by a brief description of the composting process. The benefits of composting and compost utilization are discussed, followed by a review of some of the common and innovative uses for compost. Secondly, an explanation is given of global warming, or the "greenhouse effect", and the relevance of the three GHGs measured in this project are also discussed. Finally, current literature pertaining to GHG emissions from composting is summarized.

1.2.1 Composting defined

Composting can most concisely be defined as the biological degradation of organic material under controlled, aerobic conditions (Epstein, 1997). The primary goal of composting is to convert organic waste material, including organic materials that are potentially putrescible, pathogenically infected, or otherwise hazardous, to a stable, sanitary product (Epstein, 1997). This practice has been common in agriculture for the better part of the last two centuries (Rynk, 1992), and in cruder forms perhaps as early as the origins of agriculture (Minnich, 1979). The resulting stable, humus-like material displays physiochemical properties that make it suitable as a soil conditioner in the agricultural and horticultural industries.

1.2.2 The composting process

Composting is a controlled and managed process: these important characteristics separate composting from other waste management practices such as landfilling and static storage

of manure. The rate of composting and the quality of the finished compost is dependent on a number of factors. Feedstock selection is important, and the mixing and management of feedstocks to optimize the physical and chemical properties of the compost is critical for high-rate, high-temperature composting. Among these physical and chemical properties is moisture content, oxygen availability, C:N ratio, and porosity (Golueke, 1972; Minnich, 1979; Haug, 1980).

Composting is dependent on the activity of the microbial populations within the compost that degrade and stabilize organic matter during the composting process. Moisture content and oxygen availability are considered to be the most important factors in sustaining these populations (Epstein, 1997). A moisture content of 50-60% (Stofella, 2001) provides an environment that is optimum for microbiological processes and nutrient movement. Excessive moisture limits air movement through the compost, leading to unfavorable anaerobic conditions and nutrient loss through leaching. Low moisture also decreases biological activity by inhibiting transport of the nutrients required by microorganisms for energy and subsequent biodegradation.

1.2.3 Benefits of composting and compost

Composting is unique as a waste management alternative because it yields a useable end-product, whereas traditional chemical and physical means of waste management, such as landfilling, do not (Minnich, 1979). Additionally, composting diverts organic material from the overburdened landfill system. Finished compost traditionally has been utilized mainly within the horticultural industry because it improves soil physical and chemical

characteristics and enhances crop growth. However, compost utilization has begun to diversify to include more specialized applications such as biofiltration and plant disease suppression.

1.2.4 Compost as a soil conditioner

Soil conditioning is the improvement of physical properties such as soil structure, infiltration and permeability, and susceptibility to erosion and runoff. Compost conditions soil because it is rich in organic matter. The organic matter content of compost varies widely, from as low as 30% to as high as 70% (Epstein, 1997).

Addition of compost to soil can improve its structure, a term used to describe the physical condition of the soil in its ability to form aggregates, thus promoting sufficient aeration and drainage. Compost improves structure in sandy soils by facilitating the aggregation of an otherwise coarse material, and improves the permeability of packed, waterlogged clay soils, allowing air and water to travel more freely down the profile of the soil (Minnich, 1979).

Closely related to the structure of a soil is its porosity, which describes the size and arrangement of voids within the soil matrix. The porosity of a soil affects its permeability, and hence the movement of water, nutrients and air through the soil. Changing the porosity of a soil by amending it with compost has been studied extensively by Pagliai (1981). This research showed that adding composted sludges over 2 years to a sandy loam soil resulted in an overall increase in the number and percentage of the two

smallest classes of pores, 30-50 μm diameter pores and 50-500 μm diameter pores, resulting in significant increase in soil porosity in all treatments by between approximately 8-24%. The increase in pores of these sizes is beneficial to soil and plant life because they are active in the storage and transport of water and air.

Amending soil with compost can also improve the water holding capacity of soil and increase the water available to plants. This is especially beneficial in drier climates and reduces the volume of water required for irrigation. A study by Epstein et al. (1976) found that adding 240 t ha^{-1} of sludge compost to a silt loam soil increased the soil moisture content by up to 20%. Moisture retention also increased at specific water potentials when compost was added to the soil. Increases in these soil moisture parameters reduce water stress on plants during the growing season, because water is stored and transported more efficiently.

Bulk density is another important physical parameter of soil that can be altered by adding compost. A less dense soil is desirable because lighter soils are able to retain more water, thereby increasing the plant-available water within the soil. Tester (1990) found that the addition of compost to a sandy loam soil could significantly decrease its bulk density. Tester (1990) also observed that the bulk density of the soil continued to decrease below the depth at which the compost was applied, and attributed this phenomenon to the leaching of organic carbon further down the soil profile.

1.2.5 Compost utilization in agriculture

A summary of current data (Roe, 2001) indicates that amending soil with compost may increase yield in a number of crops. Among the examples provided, results varied depending on the crop selected and application rates. For example, Hornick (1988) observed no significant differences in the yield of sweet corn between the soil amended with compost (rates of 40, 80, and 160 Mg ha^{-1} y^{-1}). However at all application rates, crop growth was enhanced as compost amendment improved soil conditions by enhancing soil nutrient content, and improved the soil's water holding capacity. Conversely, Bryan et al. (1994) observed an increase in yield and mean fruit size of summer squash when grown in soil amended with three types of municipal solid waste (MSW) compost at rates ranging from 33 to 135 Mg ha^{-1}. Despite the fact that, in many cases, compost amended soils have been shown to yield larger crops, operators of intensive production systems remain hesitant to use compost as an amendment on large scales. This hesitancy arises from the difficulty in predicting how research results will translate to field scale applications, perhaps prompting the need for field-scale research trials. Additionally, the economic viability of compost use on a commercial scale remains a point of contention. That is, can the possible benefits gained by increased crop yield through compost amendment outweigh the costs of purchase, delivery, and application of compost on a large-scale?

1.2.6 Compost and biofiltration

Biofiltration is a process in which a biologically active media is used to remove odorous, volatile organic compounds from air streams (Epstein, 1997). In addition to compost,

biofilter media may be composed of materials or mixtures of materials such as soil, peat, volcanic ash, or wood waste. Contaminants passing through the biofilter are first physically adsorbed to the compost particles and the water present in the medium. The degradation and stabilization of the compounds takes place at the sites of adsorption (Ergas, 1995), where processes such as oxidation are carried out by bacteria, fungi and actinomycetes (Stofella 2001). Like the composting process, conditions in the biofilter must be monitored to ensure that they are optimal for the removal of contaminants. Moisture content appears to be the most important parameter to be monitored, however porosity, rates of infiltration of air to be treated, and depth of the biofilter are also important. High moisture content induces anaerobic conditions in a biofilter, and low moisture content inhibits the microbial population necessary for the degradation of contaminants. Van Langenhove (1986) studied the removal of hydrogen sulfide from odorous air and suggested that the optimal moisture content for this application was 65%.

1.2.7 Compost utilization and plant disease suppression

Compost is gaining popularity in the ornamental plant industry because of its apparent ability to suppress some soil-borne plant pathogens (Stofella, 2001). Hoitink and Fahy (1992) conducted a review of a number of compost mixtures and their abilities to suppress a variety of plant diseases. They found that high C:N compost mixtures, such as those comprised of large amounts of tree bark, are the most suitable for suppressing a number of diseases, including forms of root rot, and wilting diseases. Two possible mechanisms for the suppression of plant diseases by compost have been proposed (Hoitink, 1991). The "general suppression" mechanism involves a number of

microorganisms in compost that are able to suppress disease organisms such as *Phytophthora* and *Pythium* spp. The increased biological activity resulting from the addition of compost to the soil prevents the germination of pathogenic spores. "Specific suppression" occurs when only a small number of types of microorganisms are able to suppress the pathogen in question. This type of suppression is applicable to *Rhizoctonia solani*, a type of pathogen that causes rot in ornamentals. The specificity of the suppression of *R. solani* was observed when Tuitert (1998) noted that while pathogens such as *Pythium* spp. were consistently suppressed (general suppression), the same claim could not be made for *R. solani* (specific suppression).

1.3 Greenhouse gas and global warming

Widespread concern regarding possible changes to the global environment has prompted the intensification in research and assessment of such phenomena. An area of significant interest is greenhouse gases and their role in global warming. Greenhouse gases (GHG) are normally present and play a role in the heating of the Earth's atmosphere via a process most commonly known as the "greenhouse effect". Short-wave radiation emitted by the sun passes through the atmosphere and warms the Earth's surface. Some of this energy, however, is re-emitted as longer-wave radiation into the atmosphere where it is absorbed by greenhouse gases, thus warming the atmosphere. Without the greenhouse effect, it is estimated that the earth's average temperature would be approximately 33°C less than the current global average of 15°C, thus making it impossible for life to exist.

There are four primary gases and two other families of gaseous compounds that have been identified as GHGs. These gases are carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), sulphur hexafluoride (SF_6), and the families of gases known as hydrofluorocarbons (HFCs) and perfluorocarbons (PFCs). Each gas, in a different manner, contributes to the greenhouse effect. The ability to absorb re-radiated thermal energy is referred to as "global warming potential" (GWP). Global warming potential is referenced to CO_2, the most abundant GHG, which, by definition, has a GWP of 1. CH_4 and N_2O, the other primary GHGs of interest, have global warming potentials of 21 and 310 respectively.

SF_6, HFCs and PFCs are not considered primary GHGs, but their potencies as GHGs prevent them from being discounted as insignificant species. SF_6 is the most potent GHG, with a GWP of 22,200. SF_6 is produced only by anthropogenic sources with approximately 80% of emissions arising from its use in high-voltage insulation. The remaining 20% of SF_6 emissions are thought to originate from magnesium foundries, where it is used to prevent the oxidation of magnesium (Maiss, 1996). HFCs are fully fluorinated hydrocarbons, and range in global warming potential from 190-14,800. HFCs are beginning to replace chlorofluorocarbons (CFCs) and hydrochlorofluorocarbons (HCFCs) because they are not sources of atmospheric chlorine which contribute to the catalytic destruction of ozone (Fenhann, 2000). PFCs range in global warming potential from 6,500-9,200. Despite their potencies as GHGs, little consideration is given to SF_6, HFCs, and PFCs, because they are present in very low concentrations in the atmosphere.

CO_2, CH_4, and N_2O are the most abundant GHGs and are naturally present in the earth's atmosphere at current mean concentrations of 365, 1.76, and 0.316 ppm, respectively. Increased atmospheric concentrations of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) by 31, 151, and 17% respectively over levels observed in 1750 are thought to be major contributing factors to a number of negative environmental trends with implications in regard to climate change. CO_2 is continuously cycled throughout the terrestrial and aquatic ecosystems, exchanging between soils, water, and plant life. It is thought that until the Industrial Revolution, this cycling kept atmospheric CO_2 concentrations at a relatively steady level. However, since the Industrial Revolution and the advent of fossil fuel utilization there has been a steady increase in the atmospheric concentration of CO_2. Fossil fuel combustion accounts for approximately 80% of anthropogenic CO_2 emissions. Additional activities contributing to the remaining 20% of anthropogenic CO_2 emissions include deforestation and tillage of previously undisturbed land, both of which disrupt or destroy major sinks (storage sites) of carbon.

CH_4 is produced under anaerobic conditions, with approximately 45% of the total fluxes observed originating in natural and agricultural wetlands, such as rice paddies (Milich, 1999). Additional sources of CH_4 include biomass burning, coal mining and microbial degradation of plant material, such as in the digestive process of ruminant livestock.

N_2O is emitted from a variety of natural and anthropogenic sources (Czepiel, 1995) through the processes known as nitrification and denitrification, following the pathways

described below by (1) and (2) (Barton and Atwater, 2002; Firestone and Davidson, 1989).

Nitrification:

$$NH_4^+ \rightarrow NH_2OH \rightarrow \underset{\uparrow}{\overset{NO}{[HNO]}} \rightarrow \underset{(NO_2NHOH)}{NO} \rightarrow \overset{N_2O}{\underset{\uparrow}{NO_2^-}} \rightarrow NO_3^- \quad (1)$$

Denitrification:

$$NO_3^- \rightarrow NO_2^- \rightarrow \overset{NO}{\underset{\chi}{[X]}} \rightarrow N_2O \rightarrow N_2 \qquad (2)$$

Natural sources, accounting for approximately 60% of total emissions, include undisturbed soils and oceans. Anthropogenic N_2O emissions come from cultivated soils, burning of biomass, industrial processes, such as the formation of adipic acid for use in nylon production (Fenhann, 2000), and from waste treatment, such as primary and secondary wastewater treatment (Barton and Atwater, 2002). In both natural and disturbed environments, nitrification and denitrification keep nitrogen species in steady-state equilibrium. Under moderate O_2 concentrations (~10%) (He, 2001), N_2O can "leak" from these reactions, escaping to the atmosphere.

The greenhouse effect ensures the regulation and maintenance of a habitable climate for plants and animals alike. However, elevated levels of GHGs result in an enhanced greenhouse effect, and contribute to the phenomenon known as "global warming". Recent research has suggested a direct link between increasing levels of GHGs and the increase in average global temperature. The Intergovernmental Panel on Climate Change (IPCC) has reported extensively on global warming and GHG emissions. Two reported findings of particular concern are a global average temperature increase of 0.6°C over the past century, and that the 1990s was the warmest recorded decade since reliable measurement commenced in the early 1860s. The ramifications of global warming are far-reaching, and include consequences such as increased sea levels and increased frequency of extreme weather events such as floods, droughts, cyclones and monsoons. Such environmental changes could be detrimental to the agricultural industry by decreasing crop growth and increasing animal endangerment. Risk to human life could also be increased by catastrophic weather events, infectious diseases that thrive under warmer weather conditions and air pollution (Cifuentes, 2001).

1.4. Greenhouse gas and composting

Composting has been proposed as a technology to reduce greenhouse gas emissions from waste management. Diversion of organics from land filling is thought to be "GHG friendly" because the potential production of CH_4 by landfilled organics is reduced. However, little objective data are available to validate this claim, and virtually no data are available comparing GHG emissions from composting to the landfilling of organics.

Research has shown that the primary GHGs, (CO_2, CH_4, and N_2O), are all emitted from compost. It is difficult to assess, however, how these levels compare with emissions from landfills, given the different lifecycles of each waste management strategy.

Another difficulty in the evaluation of GHG emissions from compost is the lack of standard methodologies. A concise literature review conducted by Zeman et al. (2001) reveals a need to standardize methodologies to allow accurate comparison of data and calculation of "average" GHG emission values from composting. Only then would researchers be able to evaluate the role of composting as a technology for reducing GHG emissions in waste management.

To the present, research in GHG emissions from composting has examined a wide variety of feedstocks, but a review of the available literature reveals a wide variety in analytical techniques, gas collection techniques, experimental setup, and presentation of results. This variety calls into question the validity of comparing results.

1.4.1. Review of available literature

The limited work conducted in this area allows a thorough examination and comparison of research methods and results, and makes obvious the need for standard methodologies, as suggested by Zeman (2001).

Ballestero and Douglas (1996) performed a nitrogen transport and mass-balance study to quantify the loss of nitrogen from large-scale composting facilities and determine their

impact on adjacent groundwater quality. Of particular interest was the movement of soluble N species such as ammonium (NH_4^+) and nitrate (NO_3^-) from the compost into the groundwater through runoff and leaching. However, they also considered the emission of N_2O to be a considerable pathway through which nitrogen could be lost from the systems. N_2O emissions were measured from two windrows – one composed of yard waste and the other of farm waste. Gas samples were collected from the surface of the compost using a gas flux chamber and analyzed for N_2O using a gas chromatograph equipped with an electron capture detector (GC-ECD) and a HayeSep Q polymeric packed analytical column.

The measurement and comparison of surface fluxes from both types of compost revealed a much higher N_2O emission rate from the yard waste compost than from the farm waste compost. While N_2O emissions from the farm waste remained below 1000 mg N_2O-N m^{-2} d^{-1} for the entire trial (60 days), N_2O emissions from the yard waste compost exceeded 3000 mg m^{-2} d^{-1} after 30 days, and remained above 1750 mg m^{-2} d^{-1} until the last sampling event. Ballestero and Douglas (1996) attributed this difference to higher rates of denitrification in the yard waste compost, facilitated by higher bulk density (3.10 kg m^3 for the yard waste compost; 1.40 kg m^{-3} for the farm waste compost), resulting in anoxic conditions.

Lopez-Real et al. (1996) investigated different management strategies for composting of cattle manure, and their effects on CH_4 production. The manure was handled in three different ways:

1) Minimal intervention: stockpiling manure in static piles;
2) Windrowing with scheduled turning and;
3) Windrowing with temperature-regulated forced aeration.

Gas was sampled at unspecified depths within the piles by inserting nylon tubing attached to funnels. Surface emissions were only collected for the minimal intervention pile, using a metal box (0.30 m width x 0.25 m base x 0.33 m height) and an unspecified accumulation time. Samples were collected using 5 and 10 mL gas-tight syringes and analyzed for CH_4 using a Pye Unicam gas chromatograph equipped with a flame ionization detector (GC-FID) and an analytical column packed with 80/100 Mesh Poropak Type Q.

Due to the absence of surface measurements from the turned and aerated windrows, the authors were unable to comment on CH_4 emissions. However, measurements taken at depth reflected CH_4 production rates, and it was concluded that CH_4 production was significantly higher in the minimal intervention pile than in the turned and aerated windrows. CH_4 concentrations in the minimal intervention pile ranged from 5,133-48,675 ppm, while CH_4 concentrations in the turned and aerated windrows ranged from 2.98-927 ppm and 0.96-5.95 ppm respectively. These findings provided further evidence that the aeration of compost can decrease CH_4 emissions. Furthermore, it was proposed that maintaining a frequent aeration schedule at the beginning of the composting process may be sufficient to inhibit the anaerobic bacterial community for the duration of the composting process.

Czepiel et al. (1996) quantified N_2O emissions from the composting of wastewater sludge and livestock waste. The wastewater sludge, derived from a wastewater treatment centre in Dover, New Hampshire, was mixed in a 1:1 dry weight basis with wood ash, and placed into windrows that were 18 m long, 3 m wide at the base, and 2 m high. Aeration in the windrows was provided by perforated polyethylene pipes placed underneath the compost. Aeration was computer controlled and dependent on feedback from temperature probes inserted into the compost. Composting of the livestock waste was conducted at the University of New Hampshire's Compost Research Center. The livestock waste, consisting of 25% manure and 75% bedding, was placed in windrows, measuring 11 m long, 3.7 m wide at the base, and 1.5 m high. The windrows were turned regularly with a mechanical windrow turner.

Off-gases were collected using a closed static chamber method consisting of an open collar and a detachable enclosure. Prior to sampling, the collar was placed on the surface of the compost, and allowed to equilibrate for 5-10 minutes, after which time the enclosure (51 x 51 x 26 cm) was sealed to the collar by lowering it into a water-filled channel on the top edge of the collar. A fan circulated the air in the enclosure, and five samples were drawn at 30 s intervals and analyzed using GC-ECD (Hewlett Packard Chemstation software, 2 m x 3.2 mm o.d. Poropak Q packed analytical column and 95% argon-5% CH_4 carrier gas). N_2O emissions (expressed in units of mass N_2O $area^{-1}$ $time^{-1}$), were calculated using:

$$E_{N2O} = \frac{V}{A} p \left(\frac{\Delta C}{\Delta t} \right) \qquad (3)$$

Where E_{N2O} is the emission rate, V is the volume of the enclosure (93.6 L), A is the base area (0.26 m^2) of the enclosure, p is the gas density at the temperature of the enclosure, and $\Delta C/\Delta t$ is the change in concentration of N_2O over the sampling period, obtained by linear regression of the N_2O concentration from the samples taken at 30 s intervals. Three wastewater sludge compost windrows were sampled bimonthly between April-November 1993, and two livestock compost windrows were sampled bimonthly between September-December 1992 and May-August 1993 respectively. Sampling was conducted randomly on the wastewater sludge compost windrows; however, two locations were repeatedly measured on the livestock waste compost windrows.

Being a forced aeration system, the researchers investigated whether having the aeration on or off would affect N_2O emissions. A total of 74 N_2O emission measurements were taken from the wastewater sludge compost, 33 of which were taken when the aeration system was on, and the remaining 41 with the aeration system off. Czepiel et al. (1996) found a difference in the average N_2O emission rates, depending on whether the aeration system was operating. The average N_2O emission rate when the aeration system was on was 3.4 g N_2O m^{-2} d^{-1}, while the average N_2O emission rate when the aeration system was off was 1.2 g N2O m^{-2} d^{-1}. Czepiel et al. (1996) explained that this difference can be attributed to different gas transport mechanisms at work during each of the conditions. When there is no aeration present, gases are transported through thermal convection and gaseous diffusion. However, when aeration is present, advective air movement, a much more dominant gas transport mechanism, essentially flushes trace gases such as N_2O out

of the windrow and into the atmosphere. A mass-based average was also calculated using the formula:

$$E_{N2O} = \frac{fAt}{m} \quad (4)$$

where E_{N2O} is the emission rate of N_2O, f is the average daily N_2O emission rate (g N_2O m^{-2} d^{-1}) of the entire trial, A is the effective emitting surface area (m^2) of the compost, t is the total time over which emission measurements were made (d), and m is the mass of the compost (kg). The mass-based emission rate was 0.7 g N_2O (dry kg sludge)$^{-1}$.

Czepiel et al. (1996) also attempted to find a correlation between N_2O emissions and compost age. However, they did not find any significant statistical correlation between these variables.

Surface N_2O fluxes were also measured from two livestock waste compost windrows. Using equation (3), the peak N_2O emissions were 1.3 and 0.8 g N_2O m^{-2} d^{-1} for windrows 1 and 2, respectively, and the average emission from the two piles was 0.5 g N_2O m^{-2} d^{-1}. The mass-based N_2O emission rate, calculated using equation (4) was 0.5 g N_2O (dry kg livestock compost)$^{-1}$.

The objective of a study conducted by Hellmann et al. (1997) was to profile the changing microbial communities during the composting of municipal solid waste (MSW).

However, they were also interested in quantifying CO_2, CH_4, and N_2O emissions to increase their understanding of the microbial processes occurring within a compost pile.

Hellmann et al. (1997) conducted their research on a large-scale windrow (40 m length x 2.5 m wide x 1.5 m high) consisting of 40% MSW and 60% yard waste. They used a closed chamber comprised of a polyvinyl chloride (PVC) frame and a PVC top to sample the compost off-gases. The frame remained stationary on the compost surface, but the top was attached only prior to sampling. Four samples were collected at 20 s intervals in 100 mL evacuated sample vials. The gases were analyzed using a Shimadzu 14A GC-FID/ECD.

The researchers observed high levels of CO_2 during the first 7 days, reaching a peak emission rate of approximately 360 g CO_2-C per tonne of compost per hour (t^{-1} h^{-1}), and then maintaining a relatively steady rate of between 0-100 g CO_2-C t^{-1} h^{-1} afterwards. The high emission period of CO_2 coincides with the thermophilic composting period, when microbial activity is at its peak, converting easily degradable substrates such as sugars and proteins, producing primarily water and CO_2. N_2O was detectable up until the tenth day of composting, but then in only minor amounts (i.e. approaching ambient concentrations) through the remainder of the thermophilic phase. The N_2O emission rate peaked at approximately 110 mg N_2O t^{-1} h^{-1} on day 36 of composting. The authors concluded that emissions of N_2O from compost are favored by cooler conditions that support nitrifying and denitrifying microorganisms. This explanation supports why significant N_2O emissions were only witnessed shortly after initiation (surface layers of

the compost had not yet reached thermophilic temperatures), and after day 36 (as the compost enters the low-temperature, mesophilic stage). The highest CH_4 rates were observed in the thermophilic stage, particularly between days 10 and 28 when a peak emission rate of approximately 1350 mg CH_4 t^{-1} h^{-1} was recorded. After 28 days, only negligible amounts (i.e. approaching ambient concentrations) of CH_4 were detected, leading the authors to conclude that while CH_4 may continue to be produced interstitially, methanotrophs, supported by favorable conditions in the outer regions of the compost, may consume CH_4, oxidizing it to CO_2.

Hellebrand (1998) performed experiments looking at GHG emissions from composting. In the first experiment, CH_4 and N_2O production from compost in insulated boxes was determined. Two types of compost piles were established: one was a layering of grass and soil, and the other was a layering of grass and manure. The compost mixtures were placed in boxes measuring 1.75 m high x 2 m wide x 4 m long. Each box was fitted with three polyethylene pipes at heights of 0.75 m, 1.2 m, and 1.6 m, penetrating laterally into the compost for an unspecified distance. Samples of approximately 800 mL of gas were drawn from the central region of the compost through a cooling coil to a multi-gas monitor (Bruel and Kjaer 1302). Concentrations of CH_4 and N_2O were determined on each of 6 sampling days (days 1, 4, 6, 57, 61, and 63) over 2 months.

CH_4 was evident in both treatments, but only for the first three sampling events. CH_4 production peaked for each treatment on the third sampling day; 6 days after the piles were initiated. The grass/soil pile reached a maximum CH_4 concentration of

approximately 580 ppm, while the grass/manure pile reached a maximum of approximately 150 ppm. Hellebrand (1998) attributed the higher CH_4 emissions from the grass/soil treatment to greater compaction, leading to more extensive zones of anaerobic activity. N_2O was detectable in both treatments for the duration of the trial. N_2O was greater in the grass/manure treatment, reaching a maximum of 120 ppm on the first day of sampling. The peak N_2O concentration for the grass/soil treatment was approximately 30 ppm. The higher N_2O concentrations observed in the grass/manure treatment were attributed to increased rates of nitrification/denitrification due to the higher available nitrogen content in the grass/manure treatment.

Secondly, Hellebrand (1998) conducted a laboratory-scale composting experiment to investigate the effects of ventilation rates through compost on CO_2 and N_2O emissions. The laboratory apparatus consisted of a rectangular composting vessel (0.30 m length, 0.20 m width, 0.60 m height) constructed of aluminum. Ventilation rates were controlled by a peristaltic pump, and off-gases were measured in situ via Fourier Transform Infrared Spectroscopy (FT-IR) over a period of 3 weeks. The compost feedstock selected was farm waste consisting of mixed herbage from fallow land, moistened to a moisture content of 70%. Three ventilation rates were used: 20, 50 and 100 $cm^3\ min^{-1}$. Hellebrand (1998) found that emission trends for both gases developed similarly, but differed in magnitude, depending on aeration rate. For example, CO_2 emissions peaked after approximately 24 hours for all ventilation rates, but the maximum rate was observed with the highest ventilation rate. Conversely, while N_2O emission trends were similar between the treatments (i.e. maximum emissions were observed at roughly the same time), the

average emission rates of N_2O decreased as ventilation rate increased. The researcher concluded that the higher ventilation rate allowed the compost to maintain a more aerobic environment, thus reducing the rate of denitrification which prevails under anaerobic conditions and produces N_2O.

Sommer and Dahl (1999) investigated GHG emissions from the composting of dairy cow manure. GHG emissions from pilot-scale piles of deep litter were sampled using a dynamic chamber method. The chambers, constructed of marine plywood, and mounted on a moveable steel frame, were 1.6 m high, 2 m wide and 4 m long, and were able to completely cover the pilot-scale compost heaps, which were 1.3 m high, 1.9 m wide, and 3.7 m long. The chambers had one fixed gable, and were closed on the compost pile by attaching it to a stationary gable and a fixed frame. The gable on the chambers contained an opening for incoming air, and the stationary gable had a ventilator to draw air across the surface of the compost, exiting through a steel pipe that was 2 m long and 0.4 m in diameter. Air was drawn through the opening by the ventilator and blown out the steel pipe, allowing the measurement of GHG emissions. Chambers were set on the piles 15 min. before measurements were to be taken. For the measurement of GHGs, the wind speed was set to 1.2 m s^{-1}. For each measurement, four 55 mL samples were taken with syringes at the inlet and from the steel pipe; the difference in concentration of these two measurements determined the emission rate from the compost. N_2O and CH_4 were analyzed by GC-FID/ECD (Hewlett-Packard 5890, Series II). N_2O was isolated on a 1.8 m x 3 mm analytical column packed with Porapak Q 80/100, using Ar/CH_4 (95/5% blend) was used as a carrier gas (flow rate = 30 mL min^{-1}). Temperatures of the GC

injection port and oven were 110°C and 40°C respectively, and the detector (ECD) temperature was 320°C. CH_4 was isolated on a 1.8 m x 3 mm column packed with Poropak N 80/100. Helium (He) was used as a carrier gas (flow rate = 30 mL min-1). GC temperatures remained the same for the detection of CH_4, however, the temperature of the FID detector was 270°C. CO_2 was analyzed by GC-TCD (Varian 3350). The analytical column for the isolation of CO_2 was a 2 m x 3 mm Haysep R 80/100. The carrier gas used was He (flow rate = 30 ml min^{-1}), and the temperature of the GC oven and the TCD were 30°C and 190°C, respectively. Analysis yielded concentration values for the respective gases, and emissions were calculated using the relation:

$$F = 60^2 \times 24uA(X_e - X_b) \qquad (5)$$

Where F is the emission rate (g day^{-1}), u is the air flow rate (m s^{-1}), A is the cross-sectional area of the steel pipe (m^2), and Xe and Xb are the concentrations of the gases (g m^{-3}) from the chamber, and from ambient measurements respectively.

Sommer and Dahl measured GHG emissions from three pilot-scale piles of deep litter maintained under three different management strategies: 1) compressed during storage; 2) turned once at 30 days, and; 3) untreated. Measurements were made twice on the days of initiation of the piles, once a day from days 2-6, twice a week from weeks 2-5, once a week from weeks 6-8, and once every two weeks from weeks 9-12.

CO_2 emissions were high immediately upon pile establishment, with emissions reaching 2-3 kg CO2-C per tonne of compost per day (kg t^{-1} d^{-1}). Total CO_2 emissions were highest from the pile mixed after 30 days, totaling 33 kg C t^{-1}, followed by 20 and 17 kg C t^{-1} for the compressed and untreated litter piles respectively. Total C lost, as CO_2 from the three piles was 10-19%. N_2O emissions peaked for all treatments on day 14, with the highest emission rate of 10 g N_2O-N t^{-1} from the mixed pile. Total N_2O emissions were 0.39, 0.17 and 0.04 g N t^{-1} for the untreated, compressed and mixed treatments respectively, accounting for 0.00005-0.0005% of N lost as N_2O-N from the treatments. CH_4 emissions were not detected from the untreated and compressed litter piles until 30 days after the initiation of the piles, though emissions from the turned pile were detected for the duration of the study. Peak emissions of approximately 40 g CH_4-C t^{-1} d^{-1} were observed for both the untreated and compressed litter piles on day 40.

Sommer and Moller (2000) also conducted research studying GHG emissions from compost consisting of pig litter and barley straw. The researchers varied the amount of barley straw added to the litter prior to composting, to observe how this may affect GHG emissions by creating compost recipes of different densities.

For the study, two treatments were used, 1) low amount of barley straw (bulk density = 0.44 kg L^{-1}) and; 2) high amount of barley straw (bulk density = 0.23 kg L^{-1}). The materials were formed into windrows that measured approximately 1.3 m high, 5 m long and 3 m wide. GHG emissions were measured using a closed chamber method over 130 days of composting. The circular chambers were constructed of PVC and were 0.083 m

in diameter, and 0.102 m high. Two chambers were used on each windrow; one placed on one of the side slopes, approximately 0.75 m off the ground, and one on the top of the compost pile. Gas samples were drawn just before placing the chamber on the surface, measuring ambient GHG concentrations, and at 10 and 20 minutes after the chambers were placed on the compost surface. A butyl rubber septum was fitted to the chamber to allow headspace sampling, and samples were drawn using a 60 mL syringe, and stored in 5 mL sample vials. The sampled compost off-gases were analyzed for GHGs using the same method described in earlier research conducted by Sommer and Dahl (1999).

GHG emissions were calculated using equation (6):

$$F = \frac{V}{A} \times \frac{\Delta C}{\Delta t} \quad (6)$$

Where F is the emission rate of the selected GHG (mg m^{-2} min^{-1}), V is the volume of the chamber (m^3), A is the base area of the chamber (m^2), and $\Delta C/\Delta t$ is the rate of change in gas concentration (mg m^{-3} min^{-1}) over the 20 minute sampling period.

By conducting sampling from both the side slopes of the piles, as well as the tops of the piles, Sommer and Moller (2000) were able to demonstrate the spatially variable nature of GHG emissions from compost. The rate of emission measured was greatest from the chambers located on the tops of the piles, while very low emission rates were measured from the sides or bottom of the piles. The researchers attributed this to high temperatures

within the pile creating an efflux of air through the tops of the piles, while air was drawn into the piles through the sides and bottom.

CO_2 emissions from the high-density pig litter compost were significantly ($P \leq 5\%$) higher than those observed from the low-density compost. Emissions peaked at approximately 160 and 50 g CO_2-C m^{-2} min^{-1} for the high and low density compost, respectively. Cumulative emissions from the high density compost totaled 7.37 kg CO_2-C t^{-1}, while only 0.09 kg CO_2-C t^{-1} was emitted from the low bulk density compost. The low CO_2 emission rate from the low-density compost may have been a result of the shorter time in which high temperature composting was observed. Emissions of CH_4 from the low density compost were not observed for the duration of the trial. The majority of the CH_4 emissions from the high-density compost were detectable up to approximately 30 days after the initiation of the compost pile, however there was also a small increase in emissions from days 50-60. The peak CH_4 emission rate measured from the high density compost was approximately 3.2 g CH_4-C m^{-2} min^{-1}, and the cumulative CH_4 emissions were 191.6 g CH_4-C t^{-1}. The lack of CH_4 produced by the low-density compost could be attributed to its high porosity, which limits the formation of anaerobic "pockets" where CH_4 would be produced. Like CH_4, N_2O emissions from the low-density compost was essentially undetectable, with the exception of two sampling events early in the trial where emissions reached up to 0.5 g N_2O-N m^{-2} min^{-1}. Peak emissions from the high-density compost were slightly greater, reaching approximately 0.65 g N_2O-N m^{-2} min^{-1}. The frequencies of emission events were also higher, with detectable N_2O emissions occurring until 60 days after initiation of the pile.

Cumulative emissions of N_2O were 58 and <0.1 g N_2O-N t^{-1} for the high- and low-density piles, respectively. The researchers concluded that composting pig manure of low density (i.e. with more bedding) resulted in less GHG being emitted during the composting process. They attributed this decrease to better aeration of the piles, which limited CH_4 and N_2O.

Hellebrand and Kalk (2001) measured CH_4 and N_2O from livestock waste composting using a closed-chamber method similar to that of Sommer and Moller (2000). Off-gases were sampled from a windrow of pig and cattle manure compost measuring 40 m long, 4 m wide and 1.2 m high. Seven locations were sampled from the windrow, each on a different day once per week, in order to achieve a daily resolution of emissions from the windrow. Off-gases were collected using static chambers (volume = 0.189 m^3, basal area = 0.292 m^2). 10 L sample bags were used to collect the off-gases, which were then analyzed for CH_4 and N_2O using Fourier Transform Infrared Spectrometer (FT-IR), equipped with a mercury/cadmium telluride (HgCdTe) detector and a 5 L heated measuring cell with an optical path length of 20 m. Prior to collection of compost off-gases, ambient samples were collected and analyzed. GHG emission rates were then calculated from the concentration change between the ambient air and static chambers. Emission results were presented over four composting periods, each lasting 50-70 days.

Helebrand and Kalk (2001) discovered similar time dependencies for the emissions of GHGs within each of the four periods of composting. In each composting period, CH_4 emissions increased over the first 2-3 weeks (with peak concentrations ranging from 4-12

g CH_4 m^{-2} h^{-1}), then dropped below 10% of peak levels after 2-3 weeks, and then remained low until the conclusion of the period. Hellebrand and Kalk (2001) conjectured that this was due to the establishment of anaerobic sites within the compost, supporting methane-generating bacteria. The researchers proposed that the suppression of CH_4 emissions after 2-3 weeks was due to the particular system of composting utilized at the facility, where the manure is layered on top of existing compost piles on a daily basis. The mean total emission of CH_4 from the four composting periods was 1346 g CH_4 m^{-2}. N_2O production followed a different trend, with the majority of emissions occurring approximately 3 weeks after the initiation of the trials. The mean total emission of N_2O from the four composting periods was 12.8 g N_2O m^{-2}.

He et al. (2001) studied the emissions of N_2O from composted food waste. Their study focused on determining optimal aeration rates and management strategies (e.g. intermittent feeding of compost with fresh material) in order to minimize N_2O emissions. They also studied the addition of other constituents, such as nitrite (NO_2^-), to observe whether this may inhibit or enhance N_2O emissions.

Composting was conducted on a laboratory scale, using 18 L, cylindrical compost vessels. Compost in the vessels was continuously mixed mechanically and aeration was provided by forcing air through the false floor with an air compressor. The compost was a mixture of synthetic food waste (a fortified mixture of foods of standard composition) and sawdust. In some of the composting vessels, composted cow manure was added to observe its effect on N_2O emissions. The trials lasted 150 days, with sampling occurring

every 12 hours for the first week, then every 3 days for the duration of the trial. Off-gas samples were collected by drawing air from the vessels into aluminum sample bag. Analysis of N_2O was accomplished using a Shimadzu 14A GC-ECD equipped with magnesium perchlorate pre-columns to remove water and CO_2, and a Poropak Q analytical column (2 m x 3 mm i.d. stainless steel). The carrier gas used was Ar/CH_4 (95/5%) run at a flow rate of 40 mL min^{-1}. The temperatures of the GC detector and oven were 340°C and 80°C, respectively.

In their first experiment, He et al. (2001) observed how N_2O emissions from fresh compost were affected when charged once with mature composted cattle manure (CCM). In the vessels that did not receive CCM, the majority of N_2O (>95%) production occurred 90-130 days after the beginning of the trial, with a peak concentration of approximately 10 ppm. However, in the vessels amended with CCM, N_2O production began much earlier, lasted longer, and was of greater magnitude. The majority of the N_2O from the CCM compost was produced from days 12-90, with a peak concentration of 202.67 ppm. The researchers attributed this pattern of emissions to the fact that mature compost, such as the CCM, still had the potential to be release substantial quantities of N_2O, whereas in fresh compost, significant emissions were not observed until after several weeks of composting.

Secondly, the researchers studied the effects on N_2O emissions by varying the aeration rate of the compost. Using the same 18 L laboratory vessels, the researchers chose three aerations rates (14, 56 and 70 mL min^{-1} kg^{-1}). As the aeration rate was reduced, N_2O

emissions declined significantly. These findings contradict the more commonly accepted trend of anaerobic conditions favoring denitrification and N_2O formation. However, He et al. (2001) note that lower aeration rates may have created strictly anaerobic environments, creating conditions that favor the complete reduction of NO_3^-, NO_2^- and N_2O to N_2. Data summarized by Firestone (1982) further supports this suggestion. While denitrification is an anaerobic process, as O_2 concentration is increased, changing strictly anaerobic conditions to moderately anaerobic conditions, the O_2 has an inhibitory effect on the reduction of N_2O greater than the inhibitory effect it has on other reductive processes in denitrification. Additionally, increased O_2 concentrations also allows nitrification to proceed more readily, and NH_4^+ is oxidized to NO_3^-, thereby providing the precursor for N_2O production. As a result, denitrification occurring under moderate O_2 concentrations will release more N_2O.

An experiment was also conducted to evaluate the influence of available carbon on N_2O emissions. To determine this relationship, dissolved organic carbon (DOC) and N_2O emissions were measured concurrently. The data suggested that high amounts of N_2O were produced when DOC content was low, but this was not a statistically significant correlation. This indicated that the majority of N_2O may be released after available carbon sources are depleted. The researchers were then able to successfully show how intermittent feeding of the compost with fresh food waste could prolong the primary period of N_2O production. For example, N_2O concentrations from compost which had only been amended once at the beginning of the trial began to increase at day 12 and reached a peak concentration of 202.67 ppm. However, while intermittently-charged

compost displayed very small increases in N_2O emissions each time it was amended, the primary period of production was delayed until day 47, and increased until reaching a peak concentration of 953.23 ppm on day 70. This continuous feeding of fresh waste maintains a DOC production-consumption equilibrium that inhibits ammonia oxidation (nitrification), and as long as feedstocks continue to be replenished, prolongs N_2O production.

Finally He et al. (2001), designed a short experiment to observe changes in N_2O emissions when a source of nitrite, (NO_2^-) was added to the compost. NO_2^- was added to the compost as a $NaNO_2$ solution and within 6 hours of addition, N_2O concentrations rose sharply to values of 8838.83 and 1978.96 ppm for the CCM-amended and control compost vessels, respectively. These high levels were maintained for only a short period of time (< 3 days), before returning to typical levels. These results provide evidence that N_2O is produced from NO_2^- during denitrification.

Hao et al. (2001) quantified GHG emissions from the composting of cattle feedlot manure, and investigated how varying management strategies, particularly aeration, may influence emission rates. Two treatments (performed in triplicate) were chosen for this study: 1) passive aeration, where air was introduced through the base of the compost by inserting perforated steel pipes, and; 2) active aeration, where air is introduced into the compost by mechanical turning. During the 99 day trial, the active aeration compost was turned 6 times (days 14, 21, 29, 50, 70, and 84).

Off-gases were sampled from the compost using a modified vented chamber technique, in which the chamber (0.15 m in diameter and 0.15 m high) was placed on the peak of the compost pile. Samples were then drawn from the chamber at 0, 5, 10, 20 and 30 minutes using 10 mL disposable plastic syringes. The syringe needles were capped with rubber stoppers to prevent the exchange of gases. The samples were then analyzed for GHGs using a Varian 3600 GC-FID/ECD/TCD. The analysis yielded GHG concentrations which were then plotted vs. time and fit to a second-order polynomial equation, $C = a + bx + cx^2$, where C is the concentration of gases and x is the time in minutes. Taking the derivative of this equation at time = 0 yielded the emission rate over the measured period (30 min.), and is assumed to be representative of the daily emission rate. The authors conceded that such assumptions contain a degree of uncertainty, but provide a basis for valid comparison between treatments, particularly when considering cumulative emissions.

Emissions from the passively aerated compost windrows were high for all the GHGs in the first 30 days, but declined to near ambient levels shortly thereafter. Daily emission rates varied from 0-0.22 kg CO_2-C m^{-2} d^{-1}, 0-0.04 kg CH_4-C m^{-2} d^{-1}, and 0-0.60 g N_2O-N m^{-2} d^{-1}. Cumulative emissions per tonne of manure were 73.8, 6.3, and 0.11 kg t^{-1} for CO_2-C, CH_4-C, and N_2O-N respectively. After conversion of CH_4 and N_2O emission to carbon equivalents, the total emissions from the passive treatment were 240.2 kg C t^{-1} of compost. The emission trends and magnitudes measured from the active composting windrows differed from the passive composting windrows. CO_2 and N_2O levels remained high throughout the composting trial, with the exception of two dry periods in

the middle of the experiment. Daily emission rates varied from 0-0.35 kg CO_2-C m^{-2} d^{-1} and from 0-0.92 g N_2O-N m^{-2} d^{-1}. CH_4 emissions, as in the passive composting treatment, were high initially, but then decreased, staying near ambient levels for the remainder of the trial. CH_4 emissions varied from 0-0.02 kg CH_4-C m^{-2} d^{-1}. Cumulative emissions were 168.0, 8.1 and 0.19 kg t^{-1} for CO_2, CH_4 and N_2O respectively, with the CO_2 and N_2O emissions being significantly higher than those observed from the passive aeration treatment. Total emissions from the active aeration treatment, after conversion of CH_4 and N_2O emissions to CO_2 equivalents, and the addition of machinery emissions (turner used on the treatment), were 401.4 kg C t^{-1}. Total emissions per tonne were was also significantly higher than in the passive aeration treatment.

Hao et al. (2001) attributed higher emissions of CO_2 from the active aeration treatments to high rates of biological activity, due to the maintenance of aerobic conditions facilitated by the periodic turning of the compost. Significant increases in N_2O emissions were attributed to a number of factors involving N transport and transformation. Early in the composting process, concentrations of NH_4^+ are generally much higher than NO_3^-, therefore the primary mechanism for N_2O production is nitrification. However, as time progresses, NH_4^+ concentrations decrease due to volatilization, and NO_3^- concentration increase due to nitrification. Turning, as was used in the active aeration treatment, results in a redistribution of N species, transporting the high surface concentrations of NO_3^- to the middle or bottom of the compost, and transporting higher NH_4^+ concentrations to the top. The anaerobic conditions at the bottom of a compost pile provide a suitable environment for the newly transported NO_3^- to undergo denitrification, resulting in

increased production of N_2O. An additional reason for increased emissions from the active aeration treatment is that turning mixed the compost and "fluffed" it, increasing porosity and enabling gases to diffuse more rapidly through the compost.

Beck-Friis et al. (2000) studied the emission of CH_4 and N_2O from the composting of household organic waste amended with chipped wood waste. Emissions were determined from composts of varying age, aeration pattern, and pile size. Three large windrows were used, measuring 2-2.5 m high, 5-6 m wide, and 15-20 m long. The large windrows varied in age, with emissions measurements taken from the first pile at 3 and 10 weeks, the second pile at 14 and 18 weeks, and the third pile at 26 and 40 weeks. A smaller windrow was also constructed, measuring 1.2 m high, 2 m wide, and 3 m long. Emission measurements were taken from the small windrow at weeks 1, 2, 3, 4, 6, 10 and 13. The compost was turned with a bucket loader. The larger windrows were turned at 2 and 4 week intervals, while the small windrow was turned once per week for the first two weeks, and then every other week. The trial lasted from May to October 1995.

Compost off-gases were collected using a closed static chamber method. The basal area of the chambers was 0.205 m^2, and the heights of the chambers were 0.03 and 0.25 m for the large and small compost windrows respectively. Once placed on the compost, gas samples were drawn out of a sampling port on the chamber every 15 s until five samples were collected for the small chambers, and once every minute for 5 minutes from the large chambers. The off-gases were stored in 10 mL vials until analysis.

Analysis of the compost off-gases for N_2O was conducted using a GC-ECD (Chrompak CP-9000) equipped with a Poraplot Q analytical column (25 m x 0.53 mm). The column, injector port, and detector temperatures were 38, 125, and 350 C respectively. Helium carrier and N_2 makeup gas flows were 4 and 31 mL min^{-1}. CH_4 was analyzed using a GC-FID (Packard 428) equipped with a Poropak T column.

N_2O emissions increased with compost age. N_2O emissions from the large compost pile sampled at weeks 3 and 10 ranged from 1-7 mg N_2O-N m^{-2} d^{-1}, the large compost windrow sampled on weeks 14 and 18 ranged from 9-68 mg N_2O-N m^{-2} d^{-1}, and the third pile, sampled at weeks 26 and 40, ranged from 405-1464 mg N_2O-N m^{-2} d^{-1}. The authors were brief in their discussion, linking increased N_2O emissions to compost age. A possible explanation for this relationship is that longer periods between turnings may have resulted in anaerobic conditions within the compost, supporting denitrification and N_2O production. Also, as Hao (2001) mentions, leaching of NO_3^- over time towards the moister, more anaerobic areas near the bottom of the compost could have made NO_3^- more available for denitrification. The extent to which NO_3^- leaches is dependent on its concentration profile within the pile and on the amount of moisture available in the pile to actively transport NO_3^-, which could be facilitated by rainfall if the compost were not covered. N_2O emissions from the small compost pile reached a maximum after only 3 weeks, when emissions ranged from 152-385 mg N_2O-N m^{-2} d^{-1}. A reason for the decreased emissions from the small pile could be that it was small enough to allow fresh air to move more effectively throughout the pile, thereby decreasing anaerobic zones more conducive to denitrification and N_2O production

CH_4 emissions from the large piles were highest from the compost sampled at 14 and 18 weeks, with emissions ranging from 37.4-119 g CH_4 m^{-2} d^{-1}. Emissions measured from the compost that was 3 and 10 weeks old ranged from 3.9-46.3 g CH_4 m^{-2} d^{-1} and the compost that was 26 and 40 weeks old ranged from 0.321-14.6 mg CH_4 m^{-2} d^{-1}. It may be that the highest emissions occurred in the compost aged 14-18 weeks because that is the period when O_2 concentrations are lowest, due to consumption during the thermophilic phase of composting. Compaction of the compost may have occurred, thus furthering anaerobic conditions within the pile. CH_4 emissions were much lower in the small pile, with emissions reaching a peak of 11.2 g CH_4 m^{-2} d^{-1} after 4 weeks. As in the case of N_2O emissions from the small pile, CH_4 emissions from the small pile may have been low because the smaller pile was more anaerobic.

Beck-Friis et al. (2000) also conducted an experiment measuring GHG emissions from composted household waste and chopped straw under controlled conditions in 200 L, airtight bioreactors. The researcher studied how the regulation of heating the bioreactors influenced GHG emission. Two treatments were selected and run in duplicate: 1) external heating of the compost to 55°C using temperature regulation equipment and; 2) heating of the compost through the natural composting process (i.e. "self heating"). CO_2 and N_2O emissions were measured every 5 minutes using a photoacoustic multi-gas analyzer (Innova 1312). The trials were stopped once CO_2 emissions reached less than 0.15% of initial CO_2-C per day. The external heating treatments lasted only 7 days, while the "self-heating" treatments lasted 31 and 22 days, respectively.

CO_2 emissions were much lower from the external heating treatments than the "self-heating" treatments. Emissions from the external heating treatment reached only approximately 4% of the initial C in the compost per day, and after 7 days, total C loses as CO_2 reached approximately 2-8% for the treatments. In the "self-heating" treatments, however, CO_2 measurements were substantial up to 31 days into the trial, peaking at daily emission rates of up to 17% CO_2-C of the initial C in the compost. Cumulative emissions were also much higher, totaling 65 and 67% of the initial C lost as CO_2. The authors proposed that the increased CO_2 emissions seen in the "self-heating" treatment were a result of higher rates of microbial activity resulting from a more natural composting process, whereas the external heating of the first treatment did not support thermophilic temperatures as well.

N_2O emissions were low in both treatments, with cumulative rates of 0.02% and 0.4-0.6% of total N lost as N_2O for the external and "self-heating" treatments, respectively. The cumulative values for the "self-heating" treatment are in agreement with values reported by Hellebrand (1998) and their research on GHG emissions from composted yard waste. N_2O emissions from the external heating treatment are low, compared with the "self-heating" treatment,. However, emission measurements from the external heating treatment were only recorded for 7 days, while the "self-heating" treatments were monitored for 21-33 days. It is possible that, had the external heating treatment been monitored for longer periods of time, additional emissions of N_2O may have been observed.

1.4.2 Synthesis

A review of the available literature suggests wide discrepancies in methodology, as what may be expected in the early stages of any field of research. However, one promising aspect of the current available literature is that there has been a wide variety of feedstocks used when determining GHG emissions from composting. Despite other inconsistencies, the current literature provides a base or starting point for researchers investigating numerous types of compostable material.

Also apparent throughout the literature is the range of values obtained for GHG emissions, and the difficulties in comparing the values between studies. Future research should aims to standardize of methodologies and reporting processes to aid in the comparability of data. Standard methods, merged with comparable results and the investigation of a wide variety of feedstocks, will allow compost researchers to better define GHG emission trends, and more importantly, agreed upon emission values and factors.

1.5 References

Agnew, J.M. 2002. The free air space and bulk density of compost. MSc thesis. Universtiy of Alberta, Edmonton, Alberta.

AAFC. 1999. The health of our air – Towards sustainable agriculture in Canada. H.H. Janzen, R.L. Desjardins, J.M.R. Asselin and B. Grace, Eds, Research Branch, Agriculture and Agri-Food Canada, Ottawa, ON. Electronic version available at: www.agr.ca/research/branch

Ballestero, T.P. and E.M. Douglas. 1996. Comparison between the nitrogen fluxes from composting farm wastes and composting yard wastes. Transactions of the ASAE. 39:1709-1715.

Barton, P.K. and J.W. Atwater. 2002. Nitrous oxide emissions and the anthropogenic nitrogen in wastewater and solid waste. Journal of Environmental Engineering. 128:137-150.

Beck-Friis, B., M. Pell, U. Sonensson, H. Jonsson, and H. Kirchmann. 2000. Formation and emission of N2O and CH4 from compost heaps of organic household waste. Environmental Monitoring and Assessment. 62:317-331.

Beck-Friis, B., S. Smars, H. Jonsson, and H. Kirchmann. 2001. Gaseous emissions of carbon dioxide, ammonia and nitrous oxide from organic household waste in a compost reactor under different temperature regimes. J. Agric. Eng. Res. 78:423-430.

Bryan, H.H. and C.J. Lance. 1991. Compost trials on vegetables and tropical crops. Biocycle. 32:72-77.

Cifuentes, L., V.H. Borja-Aburto, and N. Gouveia. 2001. Hidden health benefits of greenhouse gas mitigation. Science. 293:1257-59.

Czepiel, P., P. Crill, and R. Harriss. 1995 Nitrous oxide emissions from municipal wastewater treatment. Environ. Sci. Technol. 29:2352-2356.

Czepiel, P., E. Douglas. R. Harriss. and P. Crill. 1996. Measurements of N_2O from composted organic wastes. Environ. Sci. Technol. 30:2519-2525.

Epstein, E. 1997. The Science of Composting. Technomic Publiching Company, Inc. Lancaster, PA.

Ergas, S.J., E.D. Schroeder, D.P.Y. Chang, and R.L. Morton. 1995. Comtrol of volatile organic compound emissions using a compost biofilter. Water Environment Research. 67:816-821.

Fenhann, J. 2000. Industrial non-energy, non-CO2 greenhouse gas emissions. Technological Forecasting and Social Change. 63: 313-334.

Firestone, M.K. 1982. Nitrogen in agricultural soils. F.J. Stevenson, Editor. American Society of Agronomy, Madison, WI.

Firestone, M.K., and E.A. Davidson. 1989. Exchange of trace gases between terrestrial ecosystems and the atmosphere. Wiley, New York, NY.

FAO. 2001. Global estimates of gaseous emissions of NH_3, NO and N_2O from agricultural land. ISBN 92-5-104698-1. Available electronically at: www.fao.org/DOCREP/004/Y2780E/y2780e00.htm.

Golueke, C.G. 1972. Composting: A study of the process and its principles. Rodale Press, Inc. Emmaus, PA.

Hao, X., C. Chang., F.J. Larney, and G.R. Travis. 2001. Greenhouse gas emissions during cattle feedlot manure composting. J. Environ. Qual. 30:376-386.

Haug, R.T. 1980. Compost engineering: Principles and practice. Ann Arbor Science Publishers, Inc. Ann Arbor, MI.

He, Y., Y. Inamori, M. Mizuochi, H. Kong, N. Iwami and T. Sun. 2001. Nitrous oxide emissions from aerated composting of organic waste. Environ. Sci. Technol. 35:2347-2351.

Hellebrand, H.J. 1998. Emissions of nitrous oxide and other traces gases during composting of grass and green waste. J. agric. Engng. Res. 69:365-375.

Hellebrand, H.J. and W.D. Kalk. 2001. Emission of methane, nitrous oxide, and ammonia from dung windrows. Nutrient Cycling in Agroecosystems. 60:83-87.

Hellmann, B., L. Zelles, A. Palojaervi and Q. Bai. 1997. Emission of climate relevant trace gases and succession of microbial communities during open-windrow composting. Applied and Environmental Microbiology. 3:1011-1018.

Hoitink, H.A.J. and P.C. Fahy. 1986. Basis for the control of soil-borne plant pathogens with composts. Annual Review of Phytopathology. 24:93-114.

Hoitink, H.A.J., Y. Inbar, and M.J. Boehm. 1991. Status of compost-amended potting mixes naturally suppressive to soilborne diseases of florticultural crops. American Phytopathological Society. 75:869-873.

Hornick, S.B. 1988. Use of organic amendments to increase the productivity of sand and gravel spoils: Effect on yield and comparison of sweet corn. American Journal of Alternative Agriculture. 3:156-162.

IPCC. 2001. Climate change 2001: The scientific basis. United Nations Environment Programme website. Available at: http://www.grida.no/climate/ipcc_tar/wg1/006.htm.

Komilis, D.P. and R.K. Ham. 2000. A laboratory method to investigate gaseous emissions and solids decomposition during composting of municipal solid wastes. Compost Sci. Util. 8:254-265.

Lopez-Real, J. and M. Baptista. 1996. A preliminary comparative study of three manure composting systems and their influence on process parameters and methane emissions. Compost Sci. Util. 4:71-82.

M. Maiss, L.P. Steele, R.J. Francey, P.J. Fraser, R.L. Langenfelds, N.B. A. Trivett, and I. Levin. 1996. Sulfur hexafluoride - A powerful new atmospheric tracer. Atmospheric Environment. 30:1621-1629.

Milich, L. 1999. The role of methane in global warming: Where might mitigation strategies be focused? Global Environmental Change. 9:179-201.

Minnich, J. and M. Hunt. 1979. The Rodale guide to composting. Rodale Press, Inc. Emmaus, PA.

Pagliai, M., G. Guidi, M. La Marca, M. Giachetti, and G. Lucamante. 1981. Effects of sewage sludge and composts on soil porosity and aggregation. J. Environ. Qual. 10:556-561.

Roe. N.E., P.J. Stofella, and D. Graetz. 1997. Composts from various municipal solids waste feedstocks affect vegetable crops II. Growth, yields, and fruit quality. Journal of the American Society for Horticultural Science. 123:433-437.

Rynk, R. 1992. On-farm composting handbook. Northeast Regional Agricultural Engineering Services. Ithaca, NY.

Sommer, S.G. and H.B. Moller. 2000. Emission of greenhouse gases during composting of deep litter from pig production – effect of straw content. Journal of Agricultural Science. 134: 327-335.

Sommer, S.G. and P. Dahl. 1999. Nutrient and carbon balance during the composting of deep litter. J. agric. Engng. Res. 74:145-153.

Stofella, P.J., and B.A. Kahn. 2001. Compost utilization in horticultural cropping systems. Lewis Publishers, New York, NY.

Tester, C.F. 1990. Organic amendment effects on physical and chemical properties of a sandy soil. Soil Sci. Soc. Am. J. 54:827-831.

Tuitert, G., M. Szezech, and G.J. Bollen. 1998. Suppression of rhizoctonia solani in potting mixtures amended with compost made from organic household waste. Phytopathology. 88-764-773.

Zeman, C., D. Depken, and M. Rich. 2002. Research on how the composting process impacts greenhouse gas emissions and global warming. Compost Sci. Utilization 10:72.85.

CHAPTER 2

Greenhouse gas emissions from commercial composting Field Season 1: December 6, 2001 – March 7, 2002

2.1 Introduction

Composting is utilized by both the agricultural and municipal sectors as a waste management technology. Composting of organic materials such as manures, bedding, municipal solid waste and green waste (e.g. grass, tree trimmings), can result in the reduction of the total mass and volume of the wastes by as much as 50% (Rynk, 1992), and produces a stable product that can be utilized in a variety of ways.

Composting has also been put forward as a technology to reduce greenhouse gas (GHG) emissions from waste management and agriculture. It has been proposed that the diversion of organic material from landfilling reduces methane (CH_4) emissions by decreasing the amount of material undergoing anaerobic degradation. Nutrient benefits provided to plants and soil by compost amendment has the potential to decrease the need for inorganic fertilizers, which when applied can cause accumulations of nitrate (NO_3^-) in soils. This available form of N can then undergo denitrification, potentially releasing N_2O into the atmosphere (Jacinthe, P-A et al., 2000). Despite these claims, little objective data are available to support them and, of the data available, differences in methodology make the comparison of results difficult (Zeman, 2002). There is, therefore, a need for additional baseline data on GHG emissions from composting in order to more accurately quantify the potential for GHG reduction by this process.

Among the discrepancies in methodology in GHG/composting research is the way in which the compost off-gases are collected and analyzed. A number of collection methods have been used, including pumping off-gases into inert storage bags (Hellebrand and Kalk, 2001), in situ analysis of air drawn through cooling coils directly into analytical equipment (Hellebrand, 1998), and a variety of chamber methods, where gases accumulate in either closed or vented fixtures placed on the surface of the compost. The collection method used in this research was the closed, static chamber method. The advantages of the static chamber method are that the chambers are easy and inexpensive to make and maintain, can be used to measure small fluxes, and minimize site disturbance due to the short period of time they are placed on the sampling site (FAO, 2001).

The objective of this research was to collect baseline data on GHG emissions from field-scale composting, and to become familiar with the static chamber method. Concurrently, laboratory-scale composting was conducted using the same materials to evaluate the GHG emission rates from small-scale composting as a predictor for GHG emissions from large-scale composting.

2.2 Materials and methods

2.2.1 Compost establishment

Composting and GHG emission measurements were conducted over approximately three months, from December 6, 2001 to March 7, 2002. The field trial was conducted at a commercial composting site operated by Cleanit Greenit (CG) Composting Systems Inc. (20450-113 Avenue, Edmonton, AB), a fully owned and operated subsidiary of KC

Environmental Group Ltd. Laboratory- scale composting was conducted at the University of Alberta's Edmonton Research Station (Edmonton, Alberta, Canada).

Initially, one experimental windrow was established at the CG site on November 29, 2001. However, due to improper mixing of materials, the moisture content of the pile was unusually high and it froze within days. The pile was reconstructed on December 6, 2001. The CG windrow was built using a front-end loader, was trapezoidal in shape and was approximately of 12 m long, 5 m wide, and 4 m high. The pile consisted of food waste, oil sludge, kitchen grease, horse manure, construction wood waste, biosolids, and label pulp. Independent analysis of the mixed compost was not performed for this trial; however, Table 2.1 presents a summary provided by CG of the physiochemical properties of each constituent.

Table 2.1 Compost feedstock physicochemical properties

Material	C (%)	N (%)	C:N	M.C. (%)	B.D.(kg m^{-3})
Food waste	40	2.00	20	59	350
Oil sludge	57	0.20	285	59	160
Kitchen grease	38	2.40	16	69	560
Horse manure	33	0.80	41	80	800
Wood waste	54	0.18	300	30	200
Biosolid sludge	47	3.90	12	94	1000
Label pulp	37	0.07	529	50	230

Compost of a similar composition was mixed at the CG site for the laboratory-scale composting trial. The compost was transported to the University of Alberta's Edmonton Research Station and placed in four 170 L barrels. The barrels were plastic, insulated with foil-backed fibreglass, and contained grated false floors to allow leachate to drain from the compost and air to enter the compost from the bottom. No active form of

aeration was provided to the compost in the barrels. Of the four barrels, two were kept inside the building, while two were placed outside. The purpose of placing two barrels outside was to ensure that composting in the vessels took place under similar conditions to the field trial windrow. After the initiation of the field and laboratory-scale trials, the compost was not turned or watered. This was done to allow fair representation of the composting processes which take place at the CG site, where compost is produced under static conditions, with minimal intervention.

2.2.2 Compost off-gas sampling and GHG measurement using the static chamber method

Compost off-gases were collected in this experiment using a static chamber method. The chambers were completely closed and constructed of metal cylinders having an internal volume of 3.33 L and a base area of 0.0184 m^2. Variations of this method have been used in other GHG/composting research performed by Lopez-Real (1996), Czepiel et al. (1996), and Hellman (1997). Sample ports were installed by drilling holes in the tops of the chambers and fitting them with butyl septa sealed with clear silicone. Prior to sampling, a chamber was placed on the surface of the compost and gases were allowed to accumulate for 30 minutes. After 30 min., gases were sampled through the butyl septa using 30 mL Teflon syringes fitted with 38 mm (1.5")-25.5 gauge needles. 20 mL of gas were then injected into each 10 mL evacuated Exetainer® sample vial (Labco Limited, High Wycombe, United Kingdom) and stored at 4°C until analysis. Off-gases from the field-scale compost windrow were collected on days 0, 1, 6, 13, 22, 34, 41, 48, 55, 68, and 82. Emission measurements from day 22 were omitted due to experimental error. Off-gases from the laboratory-scale compost were collected on days 0, 2, 5, 16, 28, 36,

43, 57, 69, and 77. Sampling of the two outdoor vessels ceased after day 28 because the compost in the vessels was frozen and covered with a layer of ice.

Off-gases were analyzed for GHGs using gas chromatography. CO_2 and N_2O were analyzed using a gas chromatograph (GC) (Varian 3400, Varian Inc, Palo Alto, CA) equipped with an electron capture detector (ECD) and Varian Star software. Two columns were used: a Porapak N column to remove moisture, and a Haysep D (DVB solid phase, 80/100 mesh) analytical column. The temperature of the GC injector and column was set at 60°C, and the ECD temperature was set at 300°C. No temperature program was used, and the GC oven remained isothermal at 60°C. N_2 was used as the carrier gas. CH_4 was analyzed using a Varian Micro-2003 GC (Varian Inc, Palo Alto, CA) equipped with a thermal conductivity detector (TCD) and CP Maitre Elite software (Varian Inc, Palo Alto, CA). A Molesieve 5A analytical column was used, and the TCD, injector and column temperatures were all 110°C. There was no temperature program used, and the oven remained isothermal at 110°C.

GC analysis of the compost off-gases yielded parts per million (ppm) concentrations of CO_2, CH_4, and N_2O. Emission rates were calculated using a set of equations based on the ideal gas law (Laidlaw, 1993). Knowing the volume (3.33 L) and basal area (0.0184 m^2) of the static chamber, the accumulation time (i.e. the length of time the chamber was left on the emitting surface, 30 min.) and the difference between measured and ambient concentrations of the gases analyzed, the emission rate can be calculated.

The equation of state for ideal gases is:

$$PV = nRT \qquad (1)$$

Where: P = pressure (1 atm at standard pressure)
V = volume of gas (L)
n = moles of gas
R = Universal gas constant (0.08206 L atm mol-1 K-1)
T = temperature (K), 273 K at standard temperature

Rearranging (1) to isolate the number of moles (n) yields:

$$n = \frac{PV}{RT} \qquad (2)$$

Substitution of values results in:

$$n = \frac{1\,\text{atm} \times (3.33\,\text{L} \times (\text{sample gas ppm} - \text{ambient gas ppm}/1000000))}{(0.08206\,\text{L atm mol} - 1\,\text{K} - 1)(273\text{K})} \qquad (3)$$

Moles of gas (n) can then be converted into mass of gas (X) by:

$$\text{mass X} = \frac{(n \times M \times 1000000\,\mu\text{g g}^{-1})}{A} \qquad (4)$$

Where: M = molar mass (g/mol)
A = base area (m^2)

Multiplying the result of (4) (the emission rate at 30 minutes) by 2 yields an hourly flux ($\mu g\ X\ m^{-2}\ h^{-1}$).

The final emission rate is calculated by multiplying the value obtained from (4) by a temperature correction factor (F):

$$F = \frac{273}{(273 + T_A)} \qquad (5)$$

Where T_A is the ambient temperature at the time of measurement. However, in this trial, ambient temperatures were not monitored, so an ambient temperature of 0°C was assumed.

Cumulative emission rates were obtained by assuming that the hourly emission rates obtained using the preceding equations were representative of average emissions in the periods between sampling, and integrating them over the appropriate time periods (Hao, 2001). Hao (2001) concedes that, while this method of calculating daily and cumulative emissions may result in only approximate values being generated (i.e. variability between measuring periods is not accounted for), it still provides a valid method for comparison of results between trails.

Temperature monitoring was conducted prior to sampling of the off-gases by inserting temperature probes (DeMista OT-21, DeMista Instruments, Wheeling, IL, USA) into the compost to a depth of approximately 1 m near the areas at which the chambers were located.

2.3 Results

2.3.1 Temperature monitoring

Temperatures of approximately 40°C were observed in both the field and lab trials. These are bordering on the lower range of thermophilic composting. Despite remixing the compost to achieve lower moisture contents, it did not exhibit the level of activity anticipated. Additional indications of poor performance included the observation of

snow and ice remaining on the compost surface, as well as occasional sour or sulphurous odours, indicative of anaerobic conditions.

2.3.2 Field scale GHG emissions

GHG emission measurements of the field scale trial were conducted over a period of 82 days. Three static chambers were evenly spaced across the top of the compost windrow, and one chamber was placed on each of the northern and southern slopes in order to evaluate the spatial variability of emissions. Table 2.2 presents GHG emission data for each of the five chambers, detailing emission ranges and cumulative values. Chambers 1 to 3 were placed on the top of the windrow, while chambers 4 and 5 were placed on two of the slopes of the compost windrow. CO_2 emissions remained high for the duration of the trial, while CH_4 emissions increased sharply at the end of the trial. N_2O emissions were variable throughout the trial, with large emissions observed on day 6 of composting. Hourly emissions ranged from 0.36-42 g m^{-2} h^{-1} for CO_2-C, 0.46-1685 mg m^{-2} h^{-1} for CH_4-C, and 0.03-18 mg m^{-2} h^{-1} for N_2O-N.

Table 2.2: GHG Emission Data – Field Trial

Chamber	1	2	3	4	5
CO_2 Minimum (g CO_2-C m^{-2} h^{-1})	1.79	2.25	0.29	0.29	0.54
CO_2 Maximum (g CO_2-C m^{-2} h^{-1})	41.89	27.4	40.36	9.25	19.8
Cumulative (kg CO_2-C m^{-2})	9.99	2.4	9.16	1.05	1.62
CH_4 Minimum (mg CH_4-C m^{-2} h^{-1})	0.46	3.44	12.15	0.46	0.46
CH_4 Maximum (mg CO_2-C m^{-2} h^{-1})	1684.33	76.08	652.88	12.15	37.35
Cumulative (kg CH_4-C m^{-2})	0.48	0.16	0.49	0.005	0.007
N_2O Minimum (mg N_2O-N m^{-2} h^{-1})	0.06	0.23	0.05	0.04	0.03
N_2O Maximum (mg N_2O-N m^{-2} h^{-1})	13.44	12.52	1.92	0.64	17.59
Cumulative (g N_2O-N m^{-2})	1.8	0.7	0.5	0.2	1.7

Cumulative emissions ranged from 1.05-9.99 kg m^{-2} for CO_2-C, 0.16-0.49 kg m^{-2} for CH_4-C, and from 0.2-1.8 g m^{-2} for N_2O-N. Figures 2.1 to 2.6 present GHG emission profiles from each of the chambers for CO_2, CH_4 and N_2O respectively.

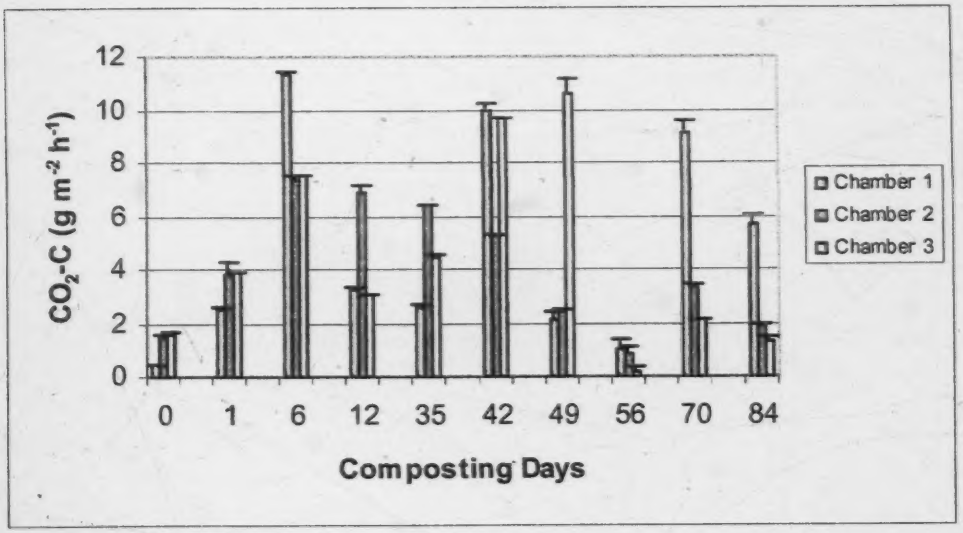

Figure 2.1: CO_2-C Emissions vs. Composting Days – Field Trial Chambers 1-3

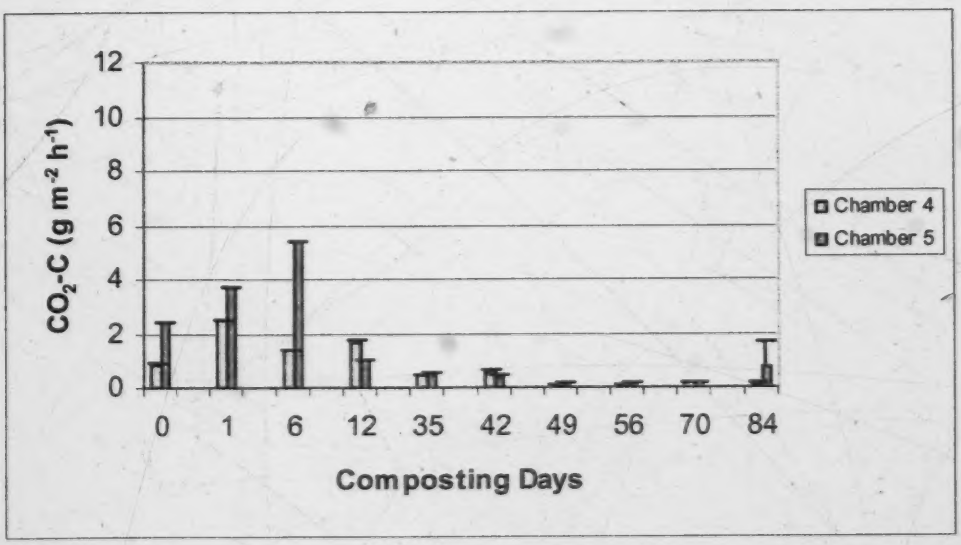

Figure 2.2: CO_2-C Emissions vs. Composting Days – Field Trial Chambers 4 and 5

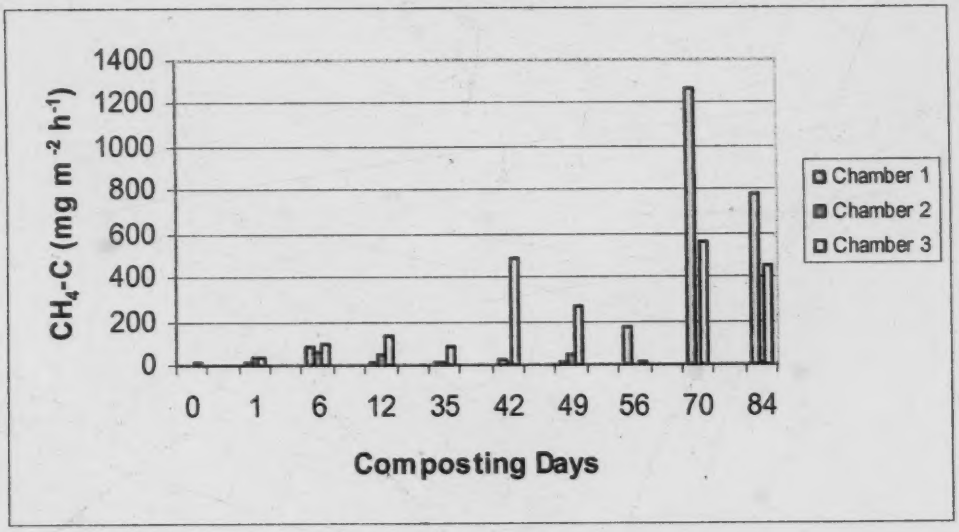

Figure 2.3: CH_4-C Emissions vs. Composting Days – Field Trial Chambers 1-3

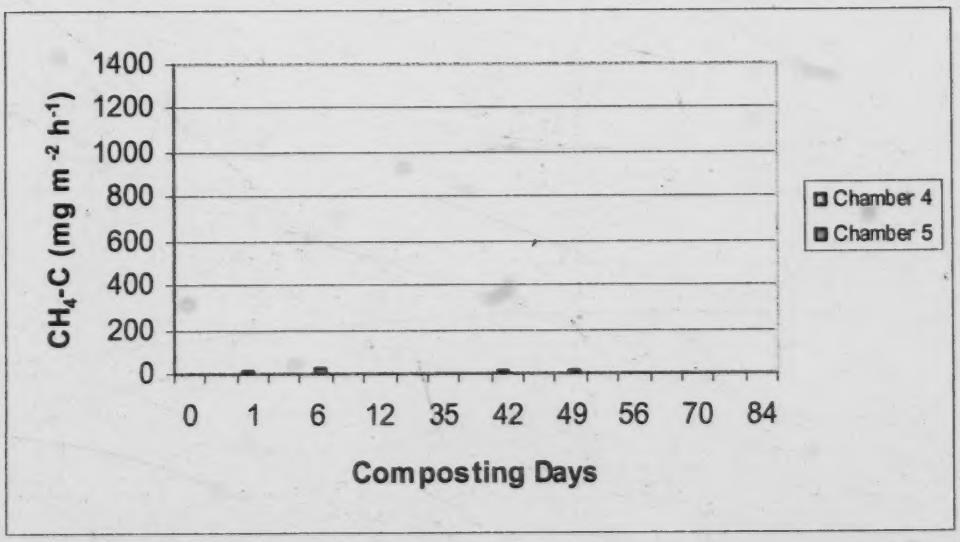

Figure 2.4: CH_4-C Emissions vs. Composting Days – Field Trial Chambers 4 and 5

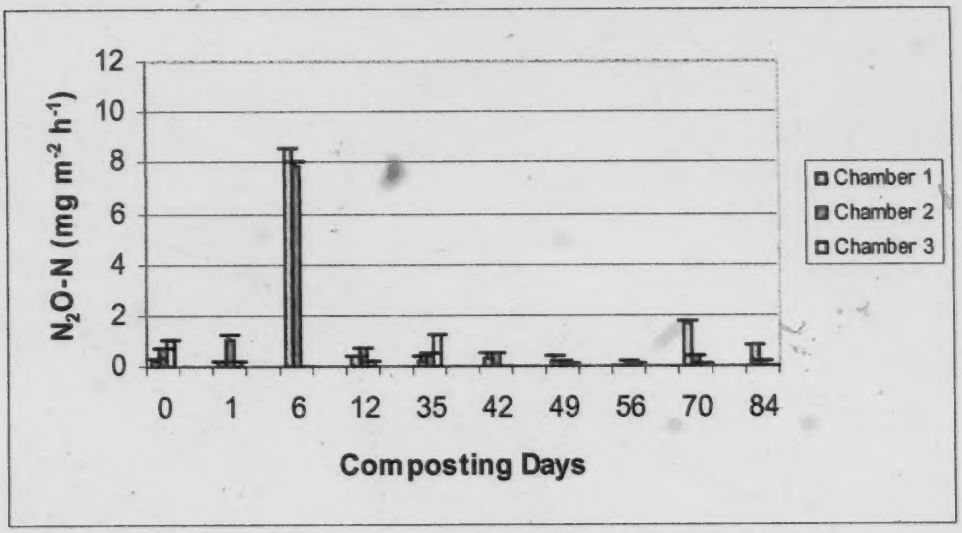

Figure 2.5: N_2O-N Emissions vs. Composting Days – Field Trial Chambers 1-3

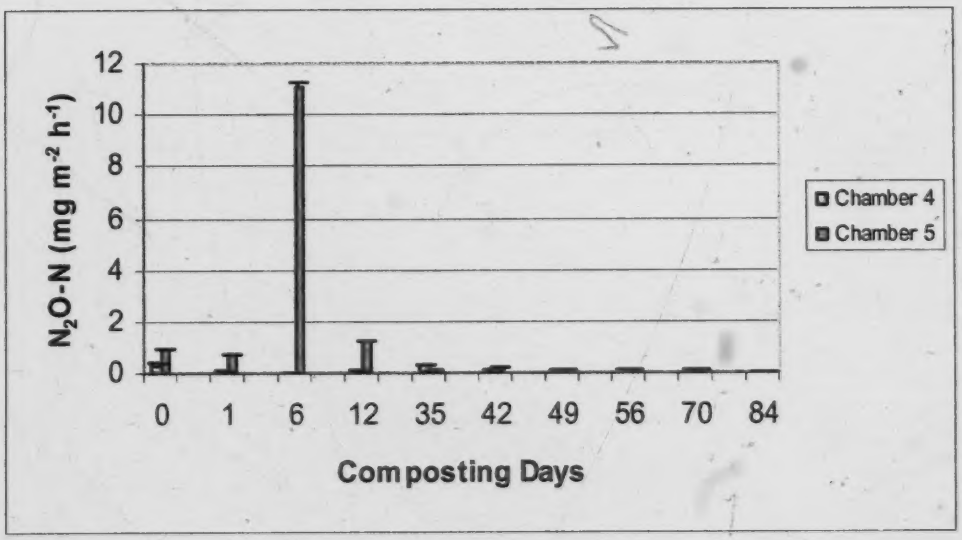

Figure 2.6: N_2O-N Emissions vs. Composting Days – Field Trial Chambers 4 and 5

2.3.3 Lab scale GHG emissions

GHG measurements from the lab scale trials were conducted over a 77 day period. One static chamber was used to collect off-gases from each vessel. Table 2.3 presents GHG emission data for each of the four vessels, detailing emission ranges and cumulative values. Chambers 1 and 2 were inside, and chambers 3 and 4 were outside. In general, CO_2 emissions were highest at the beginning of the trial, during the thermophilic stage of composting, and then decreased as the trial progressed. CH_4 emissions were highest at the beginning of the trial, and then were reduced to ambient levels by day 28. N_2O emissions, as in the field trial, peaked very early (day 5), then generally remained constant for the remainder of the trial. Hourly emissions ranged from 0.81-46 g m^{-2} h^{-1} for CO_2-C, 0.00-71 mg m^{-2} h^{-1} for CH_4-C, and 0.04-69 mg m^{-2} h^{-1} for N_2O-N.

Table 2.3: GHG Emission Data – Lab Trial

Vessel	1	2	3	4
CO_2 Minimum (g CO_2-C m^{-2} h^{-1})	2.96	3.43	1.6	0.81
CO_2 Maximum (g CO_2-C m^{-2} h^{-1})	46.08	35.07	12.8	21.78
Cumulative (kg CO_2-C m^{-2})	6.8	6.2	0.4	0.4
CH_4 Minimum (mg CH_4-C m^{-2} h^{-1})	0	0	0	0
CH_4 Maximum (mg CO_2-C m^{-2} h^{-1})	70.81	40.56	41.02	48.12
Cumulative (kg CH_4-C m^{-2})	0.007	0.003	0.0006	0.001
N_2O Minimum (mg N_2O-N m^{-2} h^{-1})	0.3	0.22	0.04	0.08
N_2O Maximum (mg N_2O-N m^{-2} h^{-1})	31.77	68.69	1.79	3.39
Cumulative (g N_2O-N m^{-2})	4.6	2.4	0.3	0.1

Cumulative emissions ranged from 0.4-6.8 kg m^{-2} for CO_2-C, 0.001-0.01 kg m^{-2} for CH_4-C, and from 0.1-4.6 g m^{-2} for N_2O-N. Figures 2.7 to 2.12 present GHG emission profiles from each of the vessels for CO_2, CH_4 and N_2O respectively.

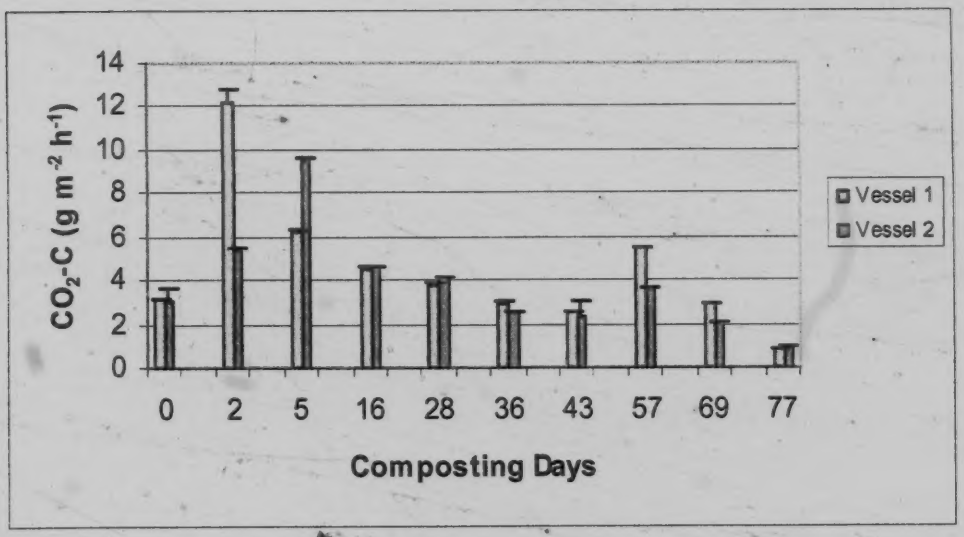

Figure 2.7: CO_2-C Emissions vs. Composting Days – Laboratory Trial Indoor Vessels

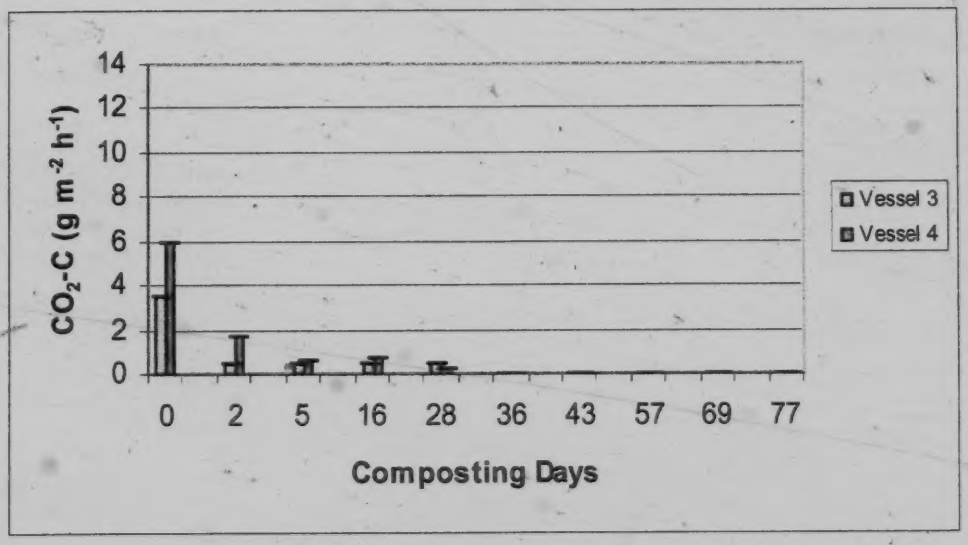

Figure 2.8: CO_2-C Emissions vs. Composting Days – Laboratory Trial Outdoor Vessels

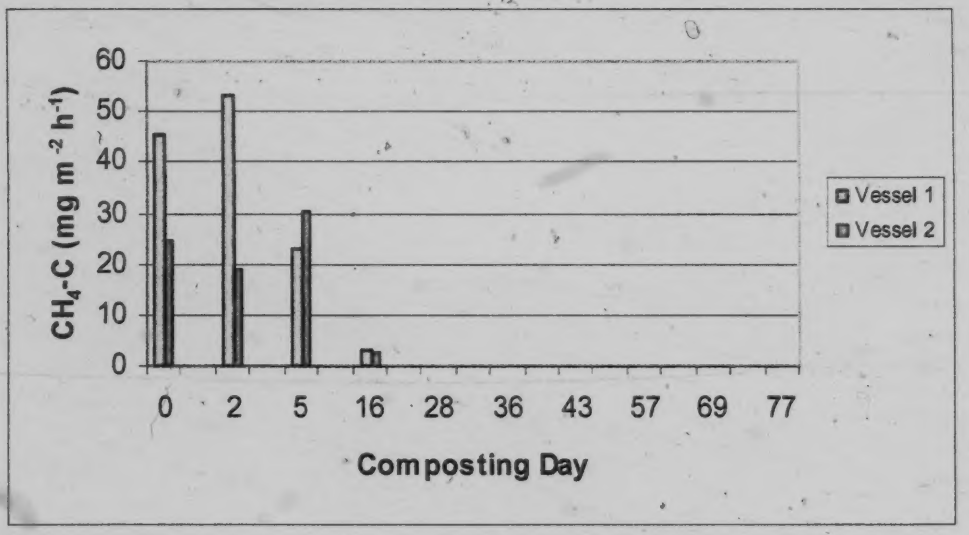

Figure 2.9: CH_4-C Emissions vs. Composting Days – Laboratory Trial Indoor Vessels

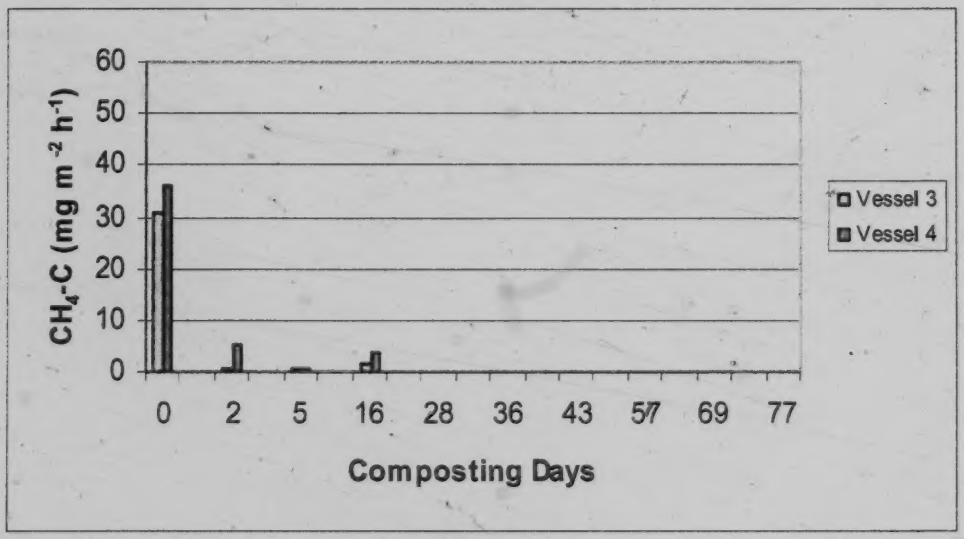

Figure 2.10: CH_4-C Emissions vs. Composting Days – Laboratory Trial Outdoor Vessels

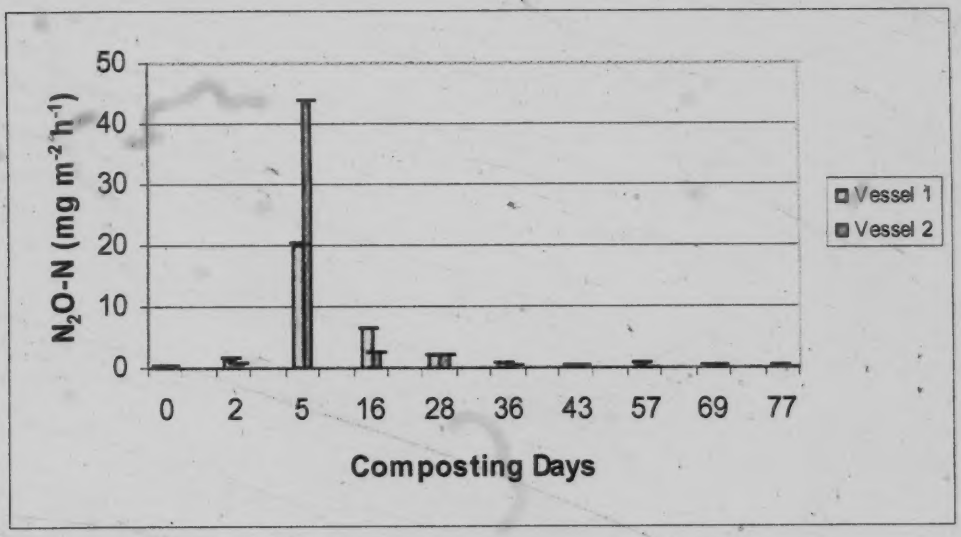

Figure 2.11: N_2O-N Emissions vs. Composting Days – Laboratory Trial Indoor Vessels

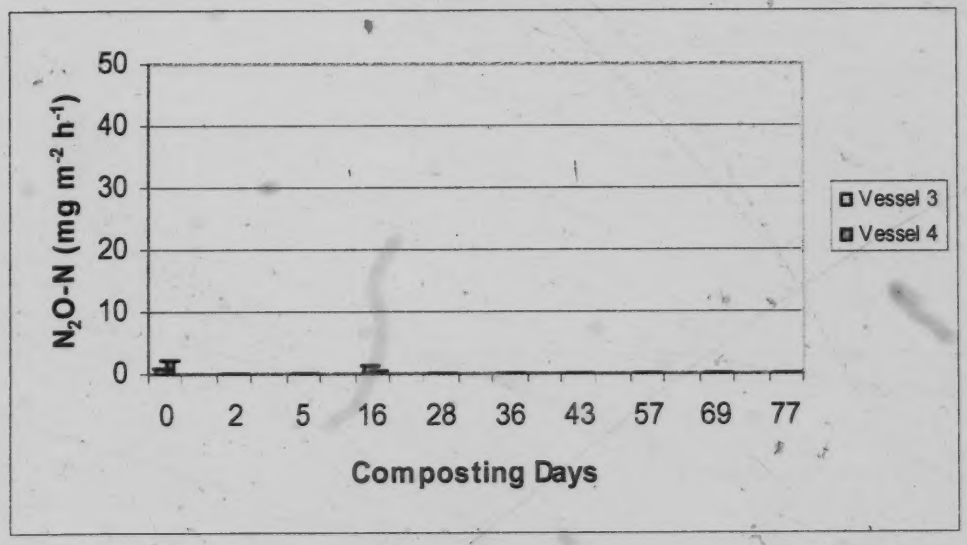

Figure 2.12: N_2O-N Emissions vs. Composting Days – Laboratory Trial Outdoor Vessels

2.4 Discussion

2.4.1 GHG emissions from field scale trial

CO_2 emissions from composting are not considered a net GHG emission since they are offset by the utilization of CO_2 for use in photosynthesis by living plant matter prior to composting (Hellmann, 1997). However, the emission of CO_2 from composting does serve as a good qualitative indicator as to how the composting process is proceeding because CO_2 and water are the primary products of the biodegradation (oxidation) of organic materials. Figure 2.1 presents the CO_2 emission data from the field trial. CO_2 emissions increased until day 6, as microbial oxidative degradation increased, and substantial emissions persisted through the remainder of the trial. Similar trends were observed in research performed by Hao (2001), who observed CO_2 emission for the duration of a 99 day trial.

With the exception of the first three measuring dates, CO_2 emissions were always higher from the top of the pile as opposed to the slopes of the windrow. Cumulative emissions of 10.0, 2.4 and 9.2 kg CO_2-C m^{-2} from the chambers located at the top of the pile compared with cumulative emissions of 1.1 and 1.6 kg CO_2-C m^{-2} from the chambers on the windrow slopes suggest that the best estimate of emissions could be obtained by considering emissions from the top of the compost exclusively. One reason proposed for the apparent spatial variability of emissions is that there is a "chimney effect" occurring due to heat generated within the windrow. Production of gases takes place throughout the pile, but the majority of the emissions escape from the top of the pile, while fresh air

is drawn into the pile through the sides and bottom of the windrow (Sommer and Moller, 2000)

CH_4 emissions remained low for the majority of the trial, however, emissions sampled from the chambers at the top of the windrow increased sharply during the last two measuring days. Between days 55 and 68, CH_4 hourly emission rates measured with the three top chambers increased from 165 to 1262, 84 to 633 and 9 to 565 mg CH_4-C m^{-2} h^{-1} respectively. This contradicts the more widely observed trend of higher CH_4 emission rates at the beginning of the trial, followed by a decrease approaching ambient concentrations (e.g. Sommer and Moller, 2000; Sommer and Dahl, 1999). In this trend, initially, during high-rate, thermophilic composting, O_2 is being consumed rapidly, resulting in the formation of anaerobic sites within the compost and increased production of CH_4. As composting proceeds, biodegradation and O_2 consumption rates decrease; as O_2 concentrations in the pile increase, aerobic conditions prevail, and CH_4 production decreases. However, a reasonable explanation for the CH_4 trend witnessed in this trial (Figure 2.3) could be based on the physical processes causing compaction and increased density of the compost windrow. Despite high O_2 consumption early in the composting process, it is possible that the compost remained porous enough to allow sufficient introduction of fresh air into the windrow to, sustain an aerobic environment. Additionally, if anaerobic conditions were to persist, any CH_4 produced in the compost may have undergone oxidation as it diffused toward the surface of the compost. However, as the trial progressed, compaction of the compost might have occurred, increasing the density of the compost and restricting the introduction of fresh air.

Furthermore, degradation of organic materials and reduction of the size of aggregates would have contributed to this effect. The resulting compacted material would result in an anaerobic environment and subsequent CH_4 production.

CH_4 emissions also appeared to show large spatial variation, with the majority of emissions being measured from the chambers at the top of the pile. Cumulative emissions from the chambers at the top of the pile were 0.48, 0.16, and 0.49 kg CH_4-C m^2, compared to only 0.005 and 0.007 kg CH_4-C m^{-2} from the chambers located on the slopes. The cumulative results from the top chambers are consistent with the findings of Hao (2001), who observed cumulative emissions of approximately 0.35 and 0.37 kg CH_4-C m^{-2} from the composting of cattle manure. These results also provide further evidence of the "chimney effect" proposed for the movement of gases through a compost windrow. Reduced CH_4 emissions from the slopes of the windrow could also be explained by considering the physical shape of the windrow. It is possible that fresh air was more easily introduced into the slopes of the compost pile either by wind, or through convection, providing aerobic conditions and reduced CH_4 production in these areas. These forces may not have been sufficient to provide enough O_2 to the middle of the compost pile, thus allowing anaerobic conditions and CH_4 production to dominate.

N_2O emissions remained relatively constant throughout the trial, with values never exceeding 2 mg N_2O-N m^{-2} h^{-1}, except for day 6, when emission rates reached 8.53, 8.20, and 11.1 mg N_2O-N m^{-2} h^{-1} for chambers 1, 2 and 5, respectively. Cumulative emissions from the chambers were 1.8, 0.7 and 0.5 g N_2O-N m^{-2} for the chambers located on the top

of the windrow, and 0.2 and 1.7 g N_2O-N m^{-2} for the chambers located on the slopes of the windrows. These values are consistent with the findings of Hao (2001), who reported cumulative N_2O emissions of 6 g N_2O-N m^{-2}, but they are much smaller than those reported by Hellebrand and Kalk (2001), who found cumulative emissions of 27.3 g N_2O-N m^{-2} from the composting of livestock manure over a 2-½ month period. Of the three primary GHGs, N_2O remains the most difficult to characterize, in terms of emission rates and general trends. The pathways by which N_2O can be produced (nitrification and denitrification), are well known, however, the large number of competing factors that govern the rates of these reactions and the extent to which N_2O may "leak" from them, is a source of much debate (Barton and Atwater, 2002). Among the factors affecting N_2O production in composting are the concentrations of the N species (including NH_4^+, NO_2^-, and NO_3^-), temperature (optimal conditions for nitrifying bacteria such as Nitrosomonas and Nitrobacter range from 5-40°C (Beck Friis et al, 2000)), moisture content, O_2 availability, and organic carbon availability.

In this experiment, the research windrow was static (i.e. not turned) and passively aerated. Under these management conditions, and based solely on the influence of oxygen availability, large emissions of N_2O could be expected. However, when considering several of the other factors, an argument for reduced N_2O emissions can be proposed. As Barton and Atwater (2002) explain, available organic carbon also plays a large part in nitrification/denitrification. When levels of NO_3^- exceed the amount of available organic carbon, incomplete utilization of NO_3^- for decomposition processes result, and N_2O is more likely to be generated in subsequent denitrification. However, in

cases where NO_3^- is limiting (i.e. an excess of organic carbon), N_2 gas dominates as the end-product, with little leakage of N_2O. It is possible that, in this study, available organic carbon levels remained high due to incomplete degradation (as witnessed by the compost's overall poor performance), and that denitrification within the pile primarily produced N_2 gas and little N_2O.

Other factors affecting denitrification are the partitioning of N species within the compost windrow and the management strategies that could influence changes in this partitioning. As Hao (2001) explains, NH_4^+ levels are high initially, and nitrification therefore dominates, resulting in increased levels of NO_3^-. Subsequent nitrification of NH_4^+ results in higher concentrations of NO_3^- near the surface, which remains stable because denitrification is less likely under the aerobic conditions near the surface. Higher concentrations of NH_4^+ in the middle and bottom of the pile also remain stable because it is less likely to undergo nitrification under the anaerobic conditions within the windrow. Management practices such as turning or watering could influence this partitioning, which in turn could lead to increased N_2O emissions. Turning could redistribute NO_3^- from the surface layers of the compost to the middle or bottom of the pile, where it could undergo denitrification under more favorable anaerobic conditions, producing N_2O. Additionally, turning of the windrow increases the porosity of the pile, thus allowing gases to more easily diffuse from the pile. Watering of piles (especially relevant in warmer climates), could cause leaching of NO_3^- to the lower portions of the compost pile, where denitrification and N_2O production is most likely to happen. Cumulative emission calculations performed by Hao (2001) revealed that N_2O emissions from an actively

aerated windrow were greater than those from a passively aerated pile, totaling 8 and 6.5 g N_2O-N m^{-2} respectively. Therefore, the low N_2O-N emissions witnessed in this trial (0.2-1.8 g N_2O-N m^{-2}), might be attributed to minimal disturbance of the piles, and the incomplete decomposition of the compost, resulting in high levels of organic carbon and the inhibition of denitrification and N_2O production.

2.4.2 GHG emissions from laboratory-scale trial

Unlike the CO_2 emission trends observed from the field windrow, emissions from the laboratory vessels behaved similarly to what has been more commonly observed in composting research; a sharp initial rise, then a gradual decrease through the course of the trial. Figure 2.4 presents the CO_2-C emission data from the laboratory trial. CO_2 emissions from vessels 1 and 2, located inside the research building, were very similar to one another, with cumulative CO_2-C emissions totaling 6.8 and 6.2 kg CO_2-C m^{-2} respectively. CO_2 emissions from vessels 3 and 4, located outside of the research building, were much lower, with each emitting a cumulative rate of only 0.4 kg CO_2-C m^{-2}. The emissions from the outside vessels were much lower because, shortly after vessel initiation, the ambient temperature dropped and the compost mass, which was not physically large enough to conserve heat, began to cool down within days. Instead of showing a gradual increase in CO_2 emissions, as in the indoor vessels, CO_2 emissions from the outdoor vessels tailed off as the compost cooled and microbial activity ceased. By day 28, ice and snow covered the outdoor vessels and sampling was stopped.

Unlike the field situation, CH_4 emissions followed trends witnessed by other researchers in emissions from composting: initially high, and then tapering off towards the end of the trial (e.g. Hellebrand and Kalk, 2001; Hellebrand, 1998; Hellmann, 1997). After only 16 days, CH_4 emissions were undetectable from all the vessels. This resulted in very low cumulative emissions, often 1-2 orders of magnitude less than emissions measured from the field-scale windrow. The mass of the compost in the vessels may have played an important role in the CH_4 emission trends. Initial CH_4 emission rates from the vessels were high, but because of the small mass of the compost, composting rates slowed, decreasing O_2 uptake and limiting the formation of anaerobic sites within the compost. Additionally, the smaller compost mass limited the amount of compaction and, therefore, sufficient porosity was maintained to sustain an aerobic environment. Also, the false floors of the vessels allowed leachate to drain from the compost. These false floors may also have supplied additional aeration to the compost, albeit passively, to help support an aerobic environment. Finally, the compost in the vessels did not perform as well as the field pile, and a dry layer was observed forming on the surface of the compost. This dry surface layer may have helped to further reduce the small amounts of CH_4 being produced in the vessels by acting as a biofilter, oxidizing to CO_2 any methane diffusing upward through the compost.

N_2O emissions were of similar magnitude to the N_2O emissions observed during the field trial. Cumulative emissions from the indoor vessels were 4.6 and 2.4 g N_2O-N m^{-2}, slightly higher than emissions measured from the field trial. N_2O emissions from the outdoor vessels were low, totaling only 0.3 and 0.1 g N_2O-N m^{-2} respectively, but this

was mainly due to the fact that GHG measurements were only taken from the outdoor vessels for the first 28 days. The emission trends of the indoor vessels and the field trial were also comparable. Generally, throughout both the field and lab trials, N_2O emission rates stayed fairly constant. However, on the third sampling day of each trial, large emission rates were observed (see Figures 3 and 6). In each case, emissions rose sharply, from an average of approximately 1 to 10 mg N_2O-N m^{-2} h^{-1} in the field trial and from approximately 2 to 30 mg N_2O-N m^{-2} h^{-1} in the lab trial. These peaks may have occurred at the time (approximately 5-6 days) when any initial NO_3^- trapped within anaerobic microsites undergo denitrification and is subsequently released as N_2O. Perhaps further substantial emissions of N_2O were not witnessed because, once initial reserves were consumed, little additional NO_3^- was introduced into anaerobic sites in the compost due to lack of turning or leaching.

N_2O emission trends appear very difficult to define; similar GHG research has yielded a variety of results and conclusions. As discussed earlier, this variability is largely due to the number of factors governing nitrification/denitrification rates and the extent to which N_2O will "leak" from these processes.

2.4.3 Laboratory-scale GHG emissions as a predictor of field-scale emissions

The purpose of measuring GHG emissions from lab-scale composting was to determine if emissions from small-scale composting could be used to predict emissions from large-scale composting. Being able to accurately predict large-scale emissions based on small-scale studies would be beneficial to the research community, as it is much easier to

conduct GHG/composting research on a lab-scale, especially if constrained by factors such as lack of space for field research. Based solely on cumulative emission results, and ignoring trends, minimum/maximum emission rates, etc., it appears that CO_2 and N_2O emissions from the lab-scale study were comparable to the emissions observed from the field. On the other hand, CH_4 emissions from the field trial were up to 2 orders of magnitude higher than those observed in the lab. In terms of emission trends, N_2O emissions were very comparable, remaining relatively constant throughout the trial, with the exception of one sampling event from which large emission events were observed. CO_2 trends differed slightly, with emissions from the field remaining high for the duration of the trial, while lab emissions peaked early and then declined gradually. CH_4 emission rates trends from the field and lab were completely different. The majority of the CH_4 emissions came near the end of the field trial, while virtually all of the CH_4 emissions from the lab trial occurred within the first three sampling days.

2.4.4 Evaluation of the static chamber method

Being a new area of research, differences in methodologies for the collection, sampling, analysis and calculation of GHG emissions from composting are apparent. An important objective of this preliminary research was to evaluate the static chamber method for use in collection of off-gases from composting.

A number of advantages and disadvantages of using closed chambers have been identified by the FAO (2001). Among the advantages listed are that closed chambers are easy to build and maintain, they are capable of measuring small emission rates, and they

provide minimal site disturbance. The chambers certainly are easy to build and maintain – the only requirements of a closed chamber is that its dimensions, basal area and volume are known, and that it is constructed of inert materials that will not absorb gases collecting in the chamber. The measurement of small emission rates was achieved in the cases of N_2O and CH_4 – ppm concentrations as low as 0.38 and 4 ppm were measurable using this technique (ambient concentration for N_2O and CH_4 are approximately 0.32 and 1.7 ppm, respectively). Site disturbance was minimized because the chambers were only placed on the surface of the compost for 30 minute intervals during the time of off-gas collection. In fact, site disturbance was more likely caused by having to walk over the surface of the compost to the specified sampling points.

Disadvantages of static chambers include the possibility of underestimation of emission rates due to the inhibition of normal emissions by the static chambers. This may happen if gases concentrate within the chamber to pressures greater than in the compost, thereby creating a pressure gradient, causing an inhibition of emissions. This problem can be alleviated by using shorter collection times. In a similar study conducted by Hao (2001), gas samples were taken over a period of 30 minutes, at intervals of 0, 5, 10, 20 and 30 minutes. Emission rates were determined by plotting a GHG concentration vs. time relationship and determining the derivative at $t = 0$. In the current study, however, off-gas samples were only taken after 30 minutes, and fluxes were calculated using the equations proposed by Laidlaw (1993). In retrospect, this lengthy accumulation time could have potentially resulted in an underestimation of flux. However, arriving at results comparable to Hao (2001) and the other limited GHG research, the extended

accumulation time did not appear to be a factor in obtaining reliable results. Hellebrand and Kalk (2001) addressed the issue of accumulation times briefly, stating that a linear approximation could reasonably be applied over a period of 10 minutes, using CO_2 as the reference gas. It is possible that this linear range could be extended beyond 10 minutes, depending on the rates at which each gas is being emitted and the dimensions of the sample chamber.

2.5 Conclusion

GHG emissions from composting were measured both on field and laboratory scales using a closed static chamber method. Emission measurements from the field windrow indicated spatial variability, with the majority of the emissions being measured from the top of the windrow, while emissions from the slopes, or sides of the windrow were essentially negligible, especially in the cases of CO_2 and CH_4. High CO_2 emissions for the duration of the trial indicate that perhaps the trial should have been extended until emission approached zero. Laboratory-scale composting using vessels at room temperature and ambient temperature yielded different results. Emissions of CO_2 and N_2O from the indoor vessels were comparable to rates observed in the field, while CH_4 emissions differed vastly, not only in magnitude (1-2 order of magnitude less than the field trials), but also in emission trends, where trends in the lab were the complete opposite of those in the field.

Based on comparison with results of other researchers, the static chamber methodology applied in this study appears to have been a valid method for the determination of GHGs

from composting. Concerns regarding overestimation of emissions of the compost off-gases due to a long accumulation time can be alleviated by decreasing the accumulation time, though it appears that a linear approximation can be applied to the accumulation of gases over the period used in this trail.

2.6 References

Barton, P.K. and J.W. Atwater. 2002. Nitrous oxide emissions and the anthropogenic nitrogen in wastewater and solid waste. Journal of Enivironmental Engineering. 128:137-150.

Beck-Friis, B., M. Pell, U. Sonensson, H. Jonsson, and H. Kirchmann. 2000. Formation and emission of N2O and CH4 from compost heaps of organic household waste. Environmental Monitoring and Assessment. 62:317-331.

Czepiel, P., E. Douglas. R. Harriss. and P. Crill. 1996. Measurements of N_2O from composted organic wastes. Environ. Sci. Technol. 30:2519-2525.

FAO. 2001. Global estimates of gaseous emissions of NH_3, NO and N_2O from agricultural land. ISBN 92-5-104698-1. Available electronically at: www.fao.org/DOCREP/004/Y2780E/y2780e00.htm.

Hao, X., C. Chang., F.J. Larney, and G.R. Travis. 2001. Greenhouse gas emissions during cattle feedlot manure composting. J. Environ. Qual. 30:376-386.

Hellebrand, H.J. 1998. Emissions of nitrous oxide and other traces gases during composting of grass and green waste. J. agric. Engng. Res. 69:365-375.

Hellebrand, H.J. and W.D. Kalk. 2001. Emission of methane, nitrous oxide, and ammonia from dung windrows. Nutrient Cycling in Agroecosystems. 60:83-87.

Hellmann, B., L. Zelles, A. Palojaervi and Q. Bai. 1997. Emission of climate relevant trace gases and succession of microbial communities during open-windrow composting. Applied and Environmental Microbiology. 3:1011-1018.

Jacinthe, P-A, W.A. Dick and L.C. Brown. 2000. Bioremediation of nitrate-contaminated shallow soils and water via water table management: evolution and release of nitrous oxide. Soil Biology and Biochemistry. 32:371-382.

Laidlaw, J.W.A. 1993. Nitrous oxide emission during the soil thaw. MSc thesis. University of Alberta.

Lopez-Real, J. and M. Baptista. 1996. A preliminary comparative study of three manure composting systems and their influence on process parameters and methane emissions. Compost Sci. Util. 4:71-82.

Minnich, J. and M. Hunt. 1979. The Rodale guide to composting. Rodale Press, Inc. Emmaus, PA.

Rynk, R. 1992. On-farm composting handbook. Northeast Regional Agricultural Engineering Services. Ithaca, NY.

Sommer, S.G. and H.B. Moller. 2000. Emission of greenhouse gases during composting of deep litter from pig production – effect of straw content. Journal of Agricultural Science. 134: 327-335.

Sommer, S.G. and P. Dahl. 1999. Nutrient and carbon balance during the composting of deep litter. J. agric. Engng. Res. 74:145-153.

Zeman, C., D. Depken, and M. Rich. 2002. Research on how the composting process impacts greenhouse gas emissions and global warming. Compost Sci. Utilization 10:72.85.

CHAPTER 3

Greenhouse gas emissions from commercial composting. Field Season 2: July 3, 2002 – September 26, 2002

3.1 Introduction

Global warming, facilitated primarily by increasing levels of GHGs, continues to be of great concern. Increases in atmospheric concentrations of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) by 31, 151, and 17%, respectively, over levels observed in 1750 (IPCC, 2001) are thought to be major contributing factors to a number of negative environmental implications with regards to climate change. The Kyoto Protocol (UNFCCC, 1992) has proposed a number of initiatives to reduce anthropogenic GHG emissions. One of particular interest identifies the need to decrease emissions, particularly CH4, from the waste management sector.

Composting has been put forward as a technology to reduce GHG emissions since it diverts organic material from anaerobic landfills. In landfills, organic material provides a substrate for the production of CH_4, which has a Global Warming Potential 21 times greater than CO_2. Despite the fact that this claim continues to be made, there is only a limited amount of objective data available on GHG emissions from composting. Literature on this topic remains sparse, and confirmation of this claim is complicated by inconsistencies in methodological approaches (Zeman, 2002). The available literature provides emission data from a wide variety of composting feedstocks, including manure composting (Hellebrand and Kalk, 2001; Hao, 2001), yard wastes (Hellebrand, 1998; Ballestero, 1996), wastewater sludge (Czepiel, 1996) and municipal solid waste (Komilis, 2000). However, the GHG collection and analysis techniques used by these authors are

as varied as the feedstocks. GHG emissions have been measured from within piles (Hellebrand, 1998,), as well as from the surface of the compost. The collection systems used vary in type, dimensions, and construction material. Chambers, both vented, and completely closed, have been used frequently (Sommer and Dahl, 1999), and samples have also been obtained by drawing air into sample bags using pumping mechanisms (He, 2001). In the case of chamber measurements, chamber dimensions range from cylinders 15 cm in diameter and 15 cm in height (Hao, 2001) to 30 cm x 25.5 cm x 33 cm rectangular boxes (Lopez-Real, 1996).

Having carried out preliminary work to establish a static chamber methodology (Chapter 2), experiments were carried out from July 3, 2003 to September 26, 2003 to test the hypothesis that high C:N ratio will suppress N_2O emissions due to the lower amount of nitrogen available for mineralization and subsequent denitrification, and that turning the compost will promote an aerobic environment in which CH_4 production is decreased.

3.2 Materials and methods

3.2.1 Compost establishment

Field measurements of GHG emissions were made on compost piles at Cleanit Greenit Composting Inc., a fully owned and operated subsidiary of KC Environmental Group Ltd., in Edmonton, AB. Four piles were established on July 3, 2002, and consisted of four treatments: 1) high C:N (45:1), turned; 2) high C:N (45:1), unturned; 3) low C:N (35:1), turned; and 4) low C:N (35:1), unturned. The piles were approximately conical in shape with a height of 3 m and a base diameter of 5 m. GHG measurements were also

taken from 170 L laboratory scale composting vessels at the University of Alberta's Edmonton Research Station using the same treatments. The compost was established in both indoors and outdoor vessels on July 4, 2002. The vessels had perforated floors supported above a plenum, but no mechanical ventilation was used. Tables 3.1 and 3.2 present the recipes used for the establishment of the low C:N and high C:N treatments, respectively.

Table 3.1 Compost feedstock physiochemical properties and recipe volumes (Low C:N Treatment)

Material	C (%)	N (%)	C:N	M.C. (%)	B.D. (kg m^{-3})	Wet tonnes	Volume (m^3)
Food waste	40	2	20	59	350	5	14.3
Oil sludge	57	0.2	285	59	160	0	0
Kitchen grease	38	2.4	16	69	560	2	3.6
Horse manure	33	0.8	41	80	800	0	0
Wood waste	54	0.18	300	30	200	5	25
Biosolid sludge	47	3.9	12	94	1000	5.5	5.5
Label pulp	37	0.07	529	50	230	0	0

C:N=31.8

Table 3.2 Compost feedstock physiochemical properties and recipe volumes (High C:N Treatment)

Material	C (%)	N (%)	C:N	M.C. (%)	B.D. (kg m^{-3})	Wet tonnes	Volume (m^3)
Food waste	40	2	20	59	350	3.5	10
Oil sludge	57	0.2	285	59	160	0	0
Kitchen grease	38	2.4	16	69	560	2	3.6
Horse manure	33	0.8	41	80	800	0	0
Wood waste	54	0.18	300	30	200	6	30
Biosolid sludge	47	3.9	12	94	1000	3.5	3.5
Label pulp	37	0.07	529	50	230	0	0

C:N=41.3

GHG emission measurements were conducted over an 84 day period. From the experience of the previous trial (Chapter 2), the size of the research windrows was decreased dramatically, allowing for less spatial variation. Also, in anticipation of the majority of emissions coming from the top of the windrow, the windrows were shaped differently – taking on a more conical shape rather than the trapezoidal shape used in the first trial. This allowed for easier sampling, as two chambers were placed side-by-side on the peak of the compost for each of the windrows.

3.2.2 Compost characterization

3.2.2.1 Determination of free air space and bulk density

Further physical characterization of the compost was conducted by measuring free air space (FAS) and bulk density (BD). Using an air pycnometer designed by and following the procedure established by Agnew (2002), FAS (%) and BD (kg m^{-3}) profiles were constructed for each of the four compost treatments from a depth of 0 m to 3 m, simulating the depth of the field piles.

3.2.2.2 Chemical analysis

Composite samples were taken from each of the four compost windrows at the CG site. Samples were dried and ground, and sent to the High Volume Laboratory (Earth Sciences Building, University of Alberta) where all chemical analysis was conducted.

Table 3.3 Summary of physicochemical methodologies

Analyte	Workup	Instrumentation	Reference
pH	1:10 compost:deionized water	Fisher Accumet 630 pH meter (Fisher Scientific, Hampton, NH, USA)	McKeague (1978)
EC	1:10 compost:deionized water	YSI Model 31 Conductivity Bridge (YSI Inc., Yellow Springs, OH, USA	McKeague (1978)
NH_4^+	1:8 compost:2M KCl sol'n	Technicon Autoanalyzer II (Technicon Industrial Systems, Tarrytown, NY, USA)	Technicon Procedure Manual (1973)
NH3	1:8 compost:2M KCl sol'n	Technicon Autoanalyzer II (Technicon Industrial Systems, Tarrytown, NY, USA)	Technicon Procedure Manual (1973)
Organic C	- 3g compost, shaken in 20 mL water - centrifugation at 10000xg for 30 min. at 4°C - analysis of filtrate (0.2 µm membrane filter)	Astro 2001 System 2 (Hach Company, Loveland CO, USA)	Hu. et al. (1972)

3.2.3 Compost off-gas sampling and GHG measurement

GHG samples were collected using a static chamber method in which a closed metal chamber (volume = 3.33 L, basal area = 0.019 m^2) fitted with a butyl septum, was placed on the surface of the compost, allowing the compost off-gases to accumulate within it. The chambers rested on the surface of the compost. Aluminum collars were left in place on the surface of the compost to reduce spatial variability in sampling. The collars were only placed on the surface of the compost to indicate sampling sites, and were not physically attached to the chambers during sampling. The accumulation time used was 20 minutes. Samples were drawn from the chambers using a 30 mL Teflon syringe fitted

with a 25.5 gauge 38mm (1.5") needle, and injected into evacuated Exetainer® sample vials. Samples were refrigerated at 4°C until analysis.

Compost off-gas samples were collected from the compost from the field trial windrows on days 0, 2, 4, 6, 8, 10, 12, 14, once per week for 6 weeks (days 21, 28, 35, 42, 49, 56) and once every two weeks for four weeks (days 70 and 84). The turned treatments were turned every other week (days 14, 28, 42, 56, 70 and 84). Off-gases from the laboratory vessels were sampled and the compost was turned on the same schedule, but only until day 56. Turning in the field was accomplished by using a front-end loader, which turned/mixed the piles for approximately 5 minutes before reforming them. In the laboratory, turning was accomplished by emptying the vessels into a large, wooden container, and turning the compost manually, using a shovel and pitchfork, for approximately 5-7 minutes. In the turned treatments, GHG samples were taken immediately after turning.

Samples were analyzed for CO_2, CH_4, and N_2O using a gas chromatograph (Varian 3400, Varian Inc., Palo Alto, CA) The GC was equipped with a flame ionization detector (FID) for the detection of CO_2 and CH_4, and an electron capture detector (ECD) for the detection of N_2O. A Porapak N precolumn was used, and a HayeSep D (DVB 80/100)/Molesieve column was used as an analytical column. The oven and injector temperature was set at 400°C, and the column temperature was isothermal at 60°C.

GHG flux was determined by converting the analyzed concentration values into emission rates (mass of elemental C or N m^{-2} h^{-1}) using the calculation method proposed by Laidlaw, 1993. Unlike the previous trial, a temperature correction factor was included in this trial, as weather data were made available. Cumulative emission rates were calculated using the same method described for the previous trial (Chapter 2).

3.2.4 Statistical analysis

Statistical analysis was conducted using the General Linear Model and Least Square Means procedures of SAS (SAS, 2001).

3.3 Results

3.3.1 Compost characteristics

3.3.1.1 Free air space and bulk density profiles

Figures 3.1 and 3.2 present Depth vs. Free Air Space and Depth vs. Bulk Density, respectively. Moisture content (MC) values are included for each treatment. The relationship between FAS and BD was also plotted for each treatment. Figure 3.3 presents an example of this relationship for the HT compost.

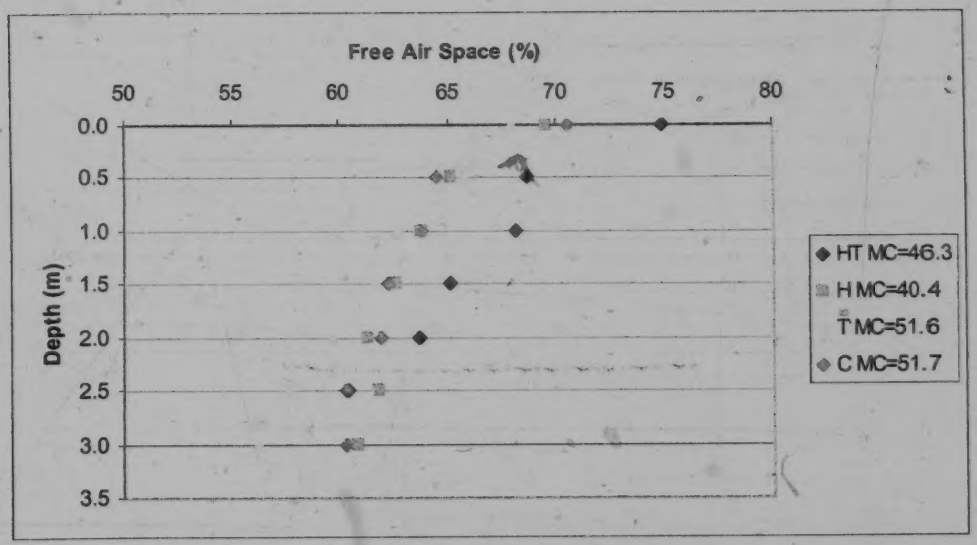

Figure 3.1: Depth (m) vs. Free Air Space (%) for HT (high C:N (45:1), turned); H (high C:N (45:1), unturned); T (low C:N (35:1), turned); and C (low C:N (35:1), unturned) treatments.

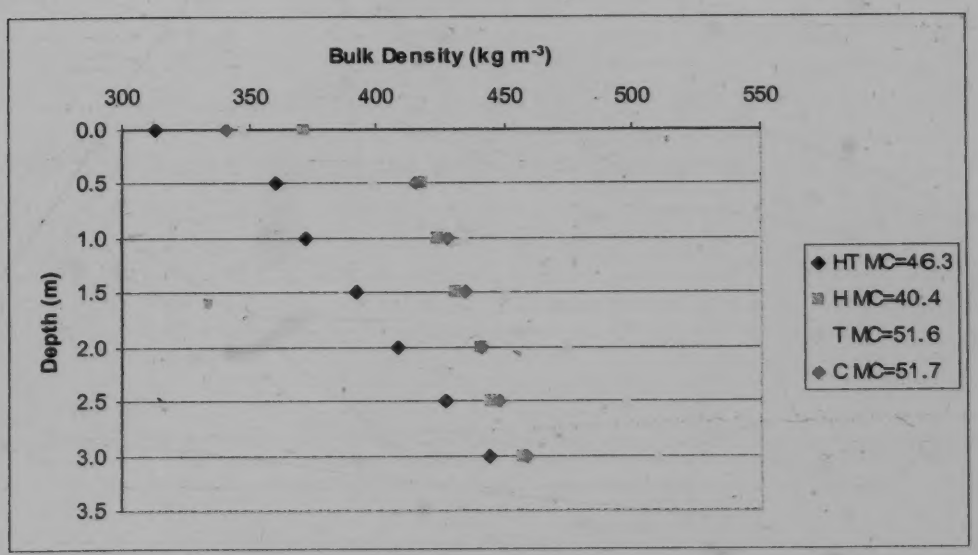

Figure 3.2: Depth (m) vs. Bulk Density (kg m^{-3}) for HT (high C:N (45:1), turned); H (high C:N (45:1), unturned); T (low C:N (35:1), turned); and C (low C:N (35:1), unturned) treatments.

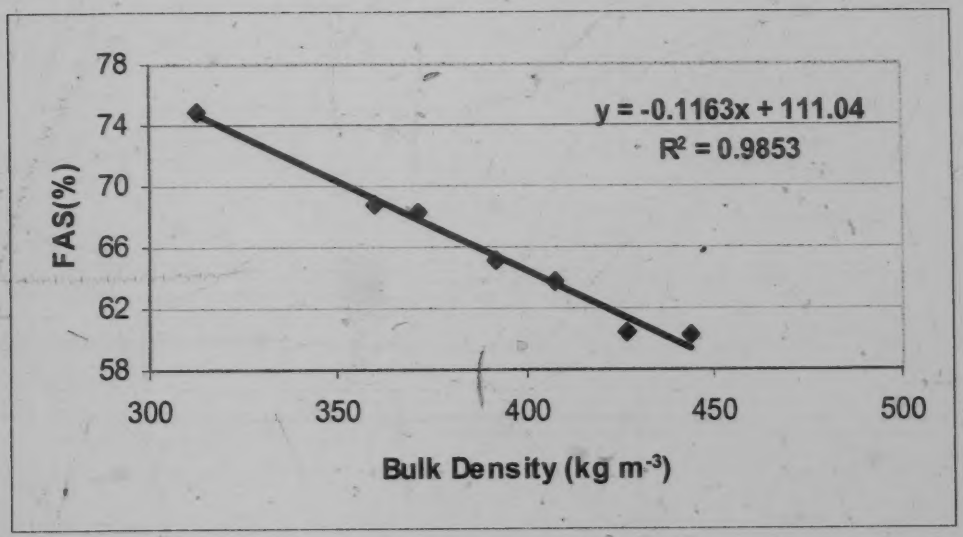

Figure 3.3: FAS (%) vs. Bulk Density (kg m-3) (High C:N/Turned Compost)

3.3.1.2 Chemical analysis

Table 3.4 presents the initial chemical analysis of the compost. Each value represents an average of 4 samples from each treatment submitted for analysis.

Table 3.4: Compost Physicochemical Analysis

Sample ID	pH	EC (ms/cm)	NO3-N (ppm in soil)	NH4-N (ppm in soil)	Soluble C (ppm in soil)	%C	%N	C:N
HT	7.00	1.18	13.23	383.81	5760.50	36.65	0.83	44.95
H	7.21	0.98	11.79	335.34	3507.00	33.52	0.76	44.10
T	7.59	0.92	7.79	154.48	4394.75	31.82	0.97	32.97
C	7.66	0.86	6.25	172.82	4930.25	32.63	0.89	37.03

HT = High C:N, turned
H = High C:N, unturned
T = Low C:N, turned
C = Low C:N, unturned

3.3.2 Field trial GHG emissions

Emissions from the two chambers used on each pile were averaged, yielding a single emission value for each GHG. Table 3.5 presents GHG emission data for each of the four treatments (HT = high C:N/turned; H = high C:N/unturned; T = low C:N/turned and; C = low C:N/unturned).

Table 3.5: GHG Emission Data – Field Trial

Treatment	HT	H	T	C
CO_2 Minimum (g CO_2-C m^{-2} h^{-1})	0	0.01	0	0
CO_2 Maximum (g CO_2-C m^{-2} h^{-1})	9.88	14.4	14.4	21.1
Cumulative (kg CO_2-C m^{-2})	3.94	4.1	3.1	6
CH_4 Minimum (mg CH_4-C m^{-2} h^{-1})	0.38	0	0	0
CH_4 Maximum (mg CH_4-C m^{-2} h^{-1})	496	1670	875	1930
Cumulative (kg CH_4-C m^{-2})	0.13	0.2	0.2	0.4
N_2O Minimum (mg N_2O-N m^{-2} h^{-1})	0	0	0	0
N_2O Maximum (mg N_2O-N m^{-2} h^{-1})	3.05	6.69	2.55	6.98
Cumulative (g N_2O-N m^{-2})	0.9	2.4	1	1.8

As in the preliminary trial (Chapter 2), CO_2 emissions remained high for the duration of the trial, and CH_4 emissions increased as composting progressed. N_2O emissions were variable for all the treatments. Cumulative emissions of CO_2 were highest for treatment C, totaling 6 kg CO_2-C m^{-2}. Treatment C also had the highest cumulative CH_4 rate at 0.4 kg CH_4-C m^{-2}, while treatment H had the highest cumulative N_2O rate at 2.4 g N_2O-N m^2. Figures 3.4 to 3.10 present GHG emission profiles from each of the field treatments.

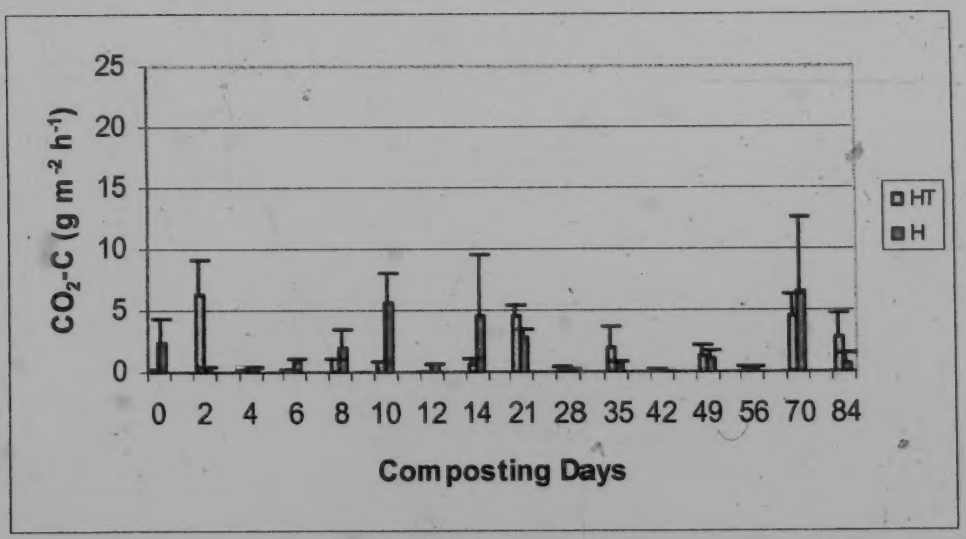

Figure 3.4: CO_2-C Emissions vs. Composting Days – High C:N/Turned (HT) and High C:N/Unturned (T) Treatments
Field Trial

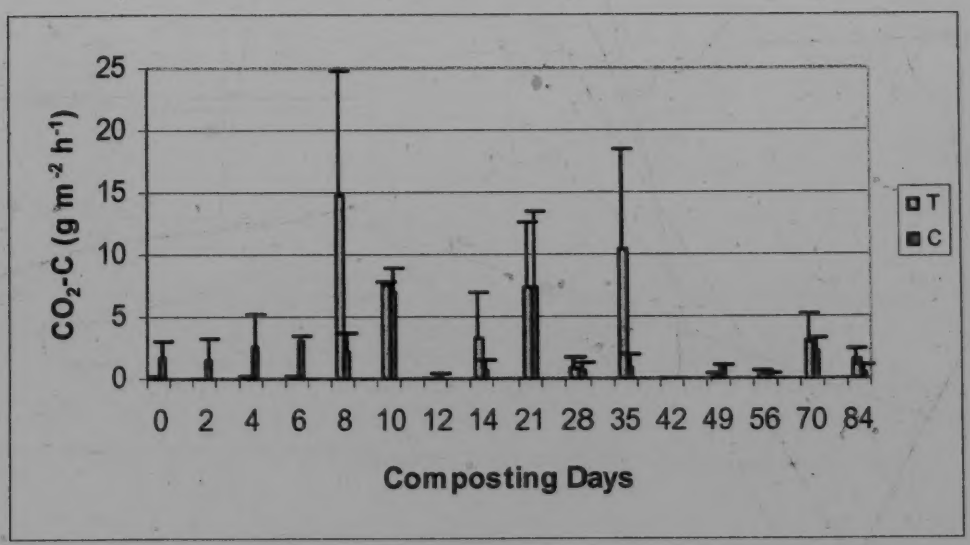

Figure 3.5: CO_2-C Emissions vs. Composting Days – Low C:N/Turned (T) and Low C:N/Unturned (C) Treatments
Field Trial

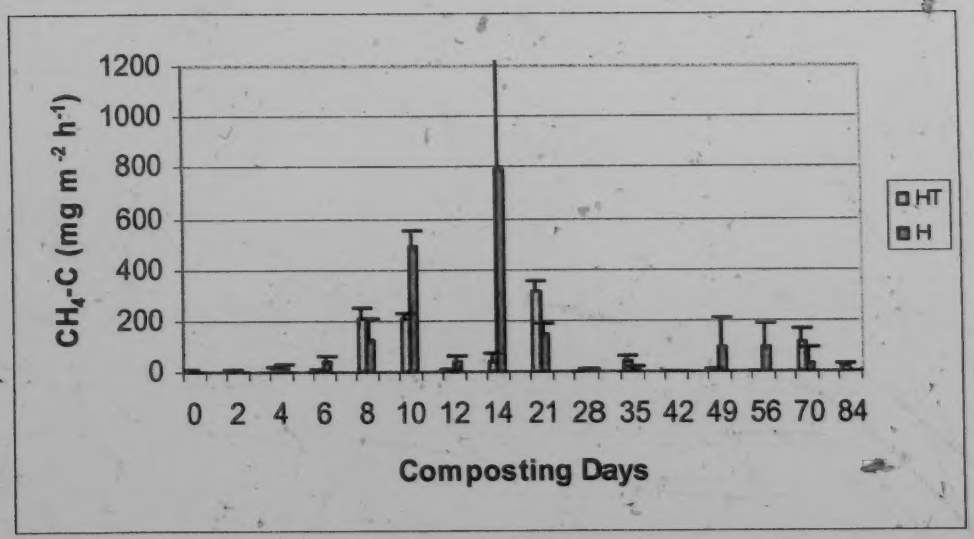

Figure 3.6: CH_4-C Emissions vs. Composting Days – High C:N/Turned (HT) and High C:N/Unturned (T) Treatments
Field Trial

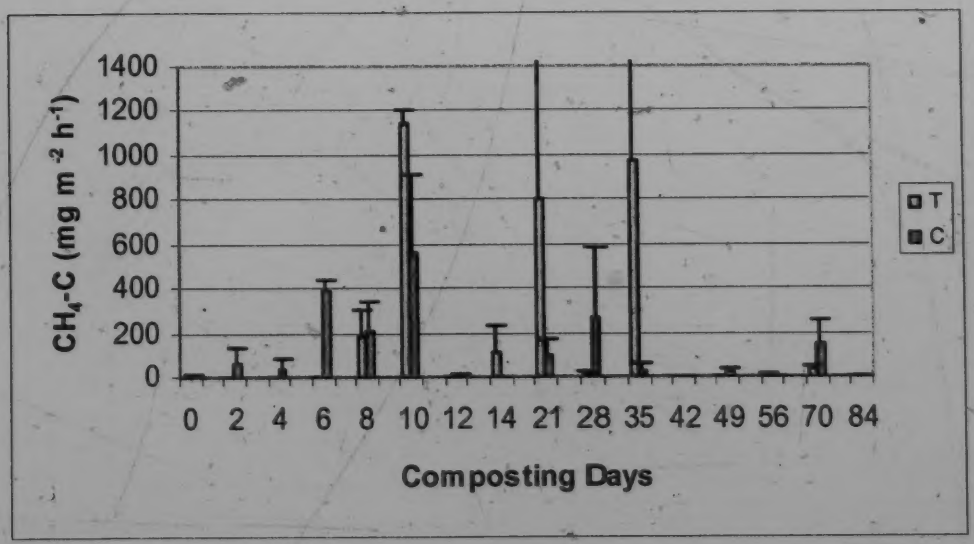

Figure 3.7: CH_4-C Emissions vs. Composting Days – Low C:N/Turned (T) and Low C:N/Unturned (C) Treatments
Field Trial

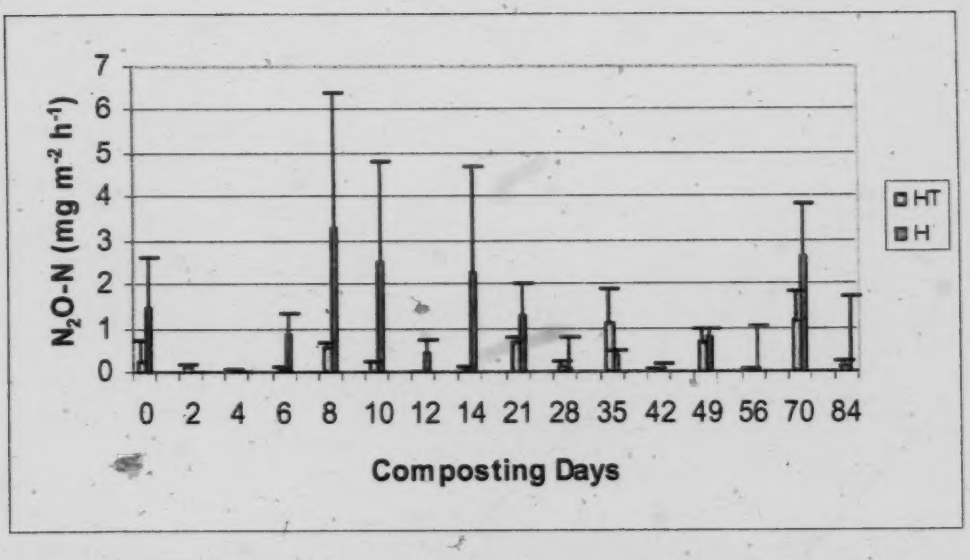

Figure 3.8: N_2O-N Emissions vs. Composting Days – High C:N/Turned (HT) and High C:N/Unturned (T) Treatments Field Trial

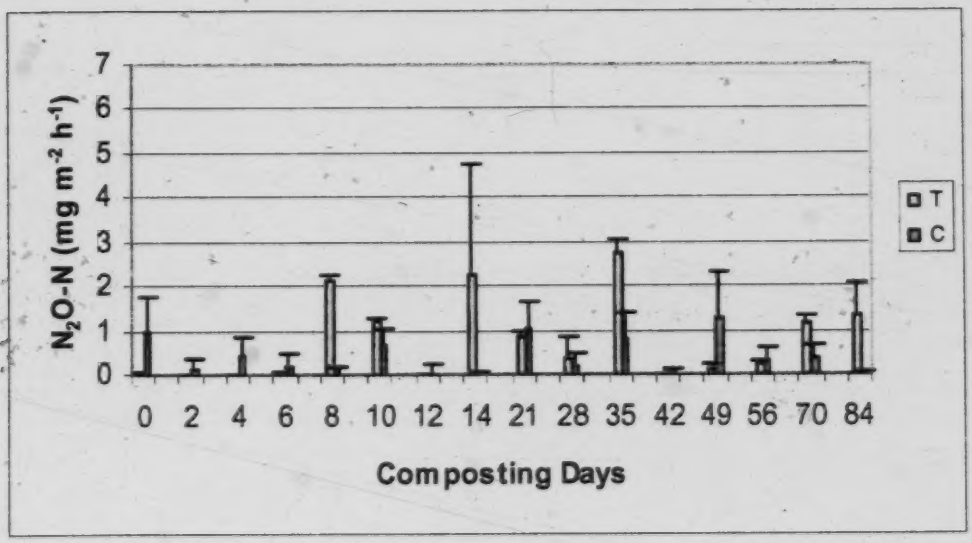

Figure 3.9: N_2O-N Emissions vs. Composting Days – Low C:N/Turned (T) and Low C:N/Unturned (C) Treatments Field Trial

3.3.3 Laboratory scale GHG emissions

GHG measurements from the lab scale trials were conducted over a 56 day period. Samples were taken over the entire 84 day period as in the field trial, however, files containing the later results were corrupted and lost after the off-gas samples had been discarded. One static chamber was used to collect off-gases from each vessel. Table 3.6 presents GHG emission data for each of the eight vessels, detailing emission ranges and cumulative values (HT = high C:N/turned; H = high C:N/unturned; T = low C:N/turned and; C = low C:N/unturned; I = indoor vessels and; O = outdoor vessel).

Table 3.6: GHG Emission Data – Lab Trial

Treatment	HT(I)	H(I)	T(I)	C(I)	HT(O)	H(O)	T(O)	C(O)
CO_2 Minimum (g CO_2-C m^{-2} h^{-1})	0.57	0.86	0.59	0.67	0	0.03	0	0.04
CO_2 Maximum (g CO_2-C m^{-2} h^{-1})	9.13	17.4	9.11	8.16	6.11	7.54	9.75	5.24
Cumulative (kg CO_2-C m^{-2})	4.1	5.3	4.7	3.3	1	1.3	1.6	0.9
CH_4 Minimum (mg CH_4-C m^{-2} h^{-1})	0	0.07	0.46	0.86	0	0	0	0
CH_4 Maximum (mg CO_2-C m^{-2} h^{-1})	300	27.8	105	22.9	3	17.7	98.5	40.1
Cumulative (kg CH_4-C m^{-2})	0.02	0.004	0.009	0.005	0.001	0.002	0.007	0.005
N_2O Minimum (mg N_2O-N m^{-2} h^{-1})	0	1.54	0	0	0	0	0	0
N_2O Maximum (mg N_2O-N m^{-2} h^{-1})	9.63	40.5	9.09	2.7	4.79	7.69	7.71	2.96
Cumulative (g N_2O-N m^{-2})	3.5	15.6	2.6	1.5	1.5	2.5	1.8	0.4

Emissions, particularly in the cases of CO_2 and CH_4, were similar to those observed in the first trial (Chapter 2). CO_2 emissions were highest at the beginning of the trial, during the thermophilic stage of composting, and then decreased as the trial progressed. CH_4 emissions were highest at the beginning of the trial, and then dropped sharply after approximately 10 days. N_2O emissions from the indoor vessels were did not fluctuate substantially, while emissions from the outdoor vessels were more variable. Cumulative emissions of CO_2 per unit area were greater from the indoor vessels than the oudoor

vessels; the highest being the indoor treatment H, totaling 5.3 kg CO_2-C m^{-2}. Cumulative CH_4 emissions were highest from the indoor treatment HT at 0.002 kg CH_4-C m^{-2}, and cumulative N_2O emissions were highest from the indoor treatment H totaling 15.6 g N_2O-N m^{-2}. Figures 3.10 to 3.15 present GHG emission profiles from each of the vessels for CO_2, CH_4 and N_2O respectively.

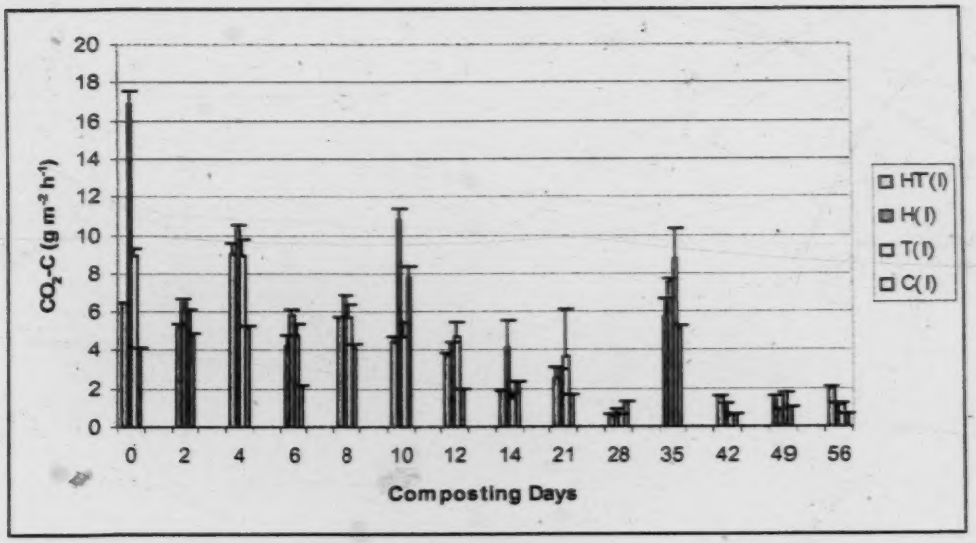

Figure 3.10: CO_2-C Emissions vs. Composting Days – Indoor Vessels Laboratory Trial

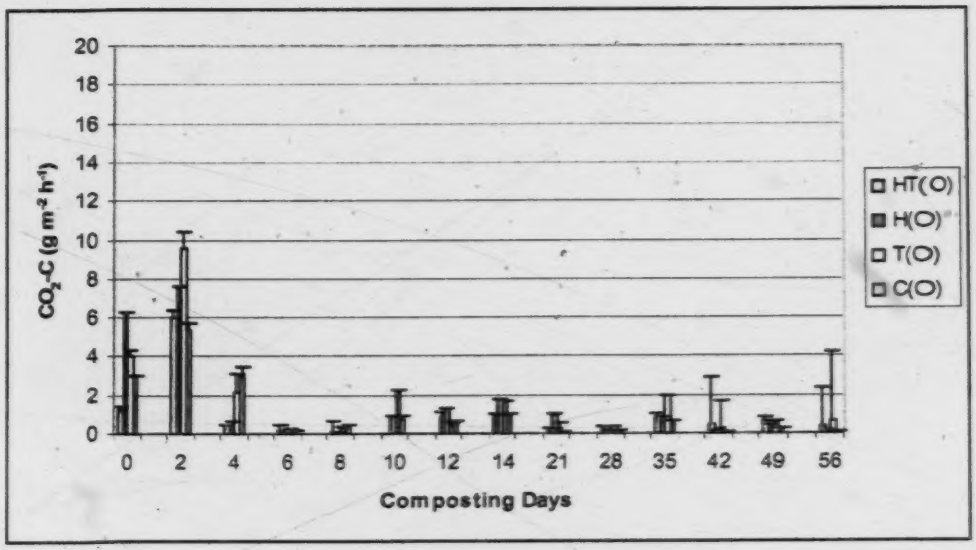

Figure 3.11: CO_2-C Emissions vs. Composting Days – Outdoor Vessels Laboratory Trial

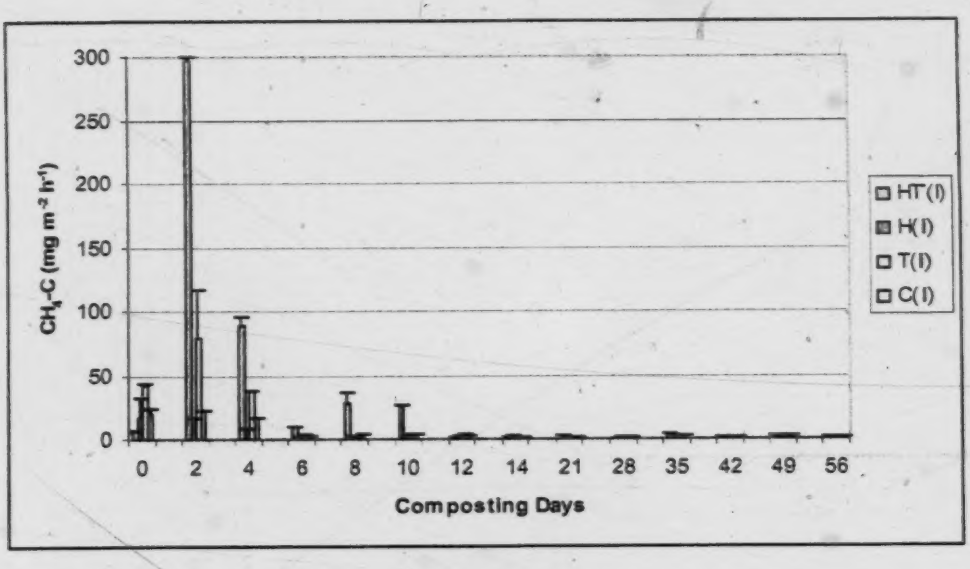

Figure 3.12: CH_4-C Emissions vs. Composting Days – Indoor Vessels Laboratory Trial

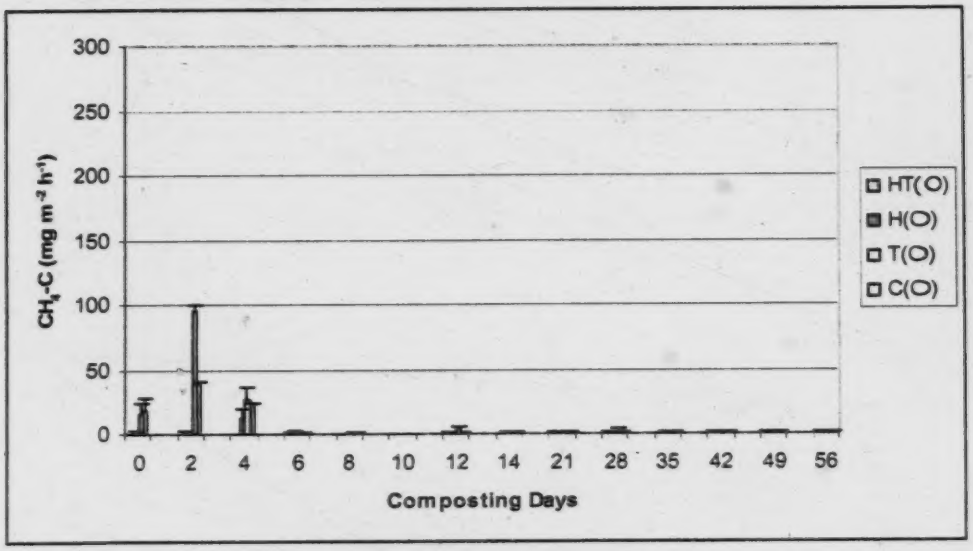

Figure 3.13: CH_4-C Emissions vs. Composting Days – Outdoor Vessels Laboratory Trial

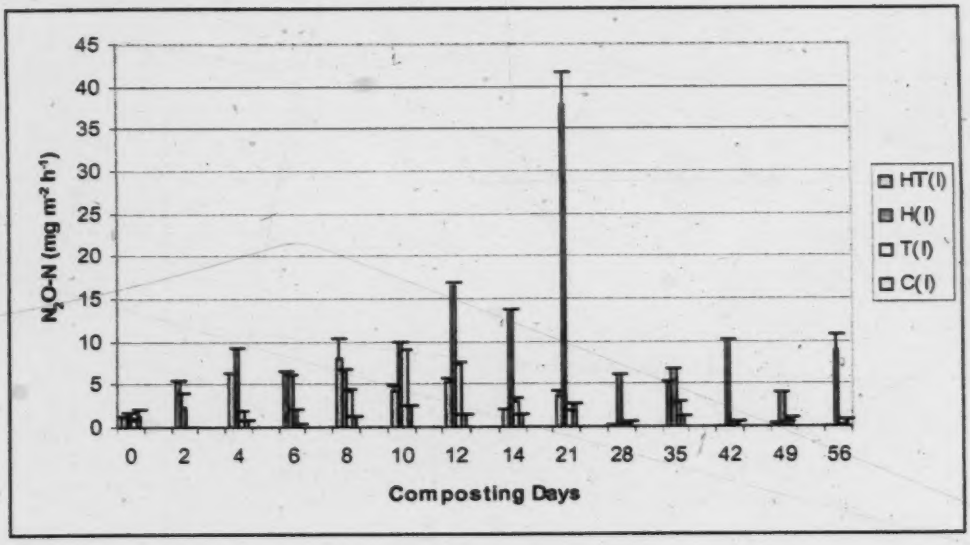

Figure 3.14: N_2O-N Emissions vs. Composting Days – Indoor Vessels Laboratory Trial

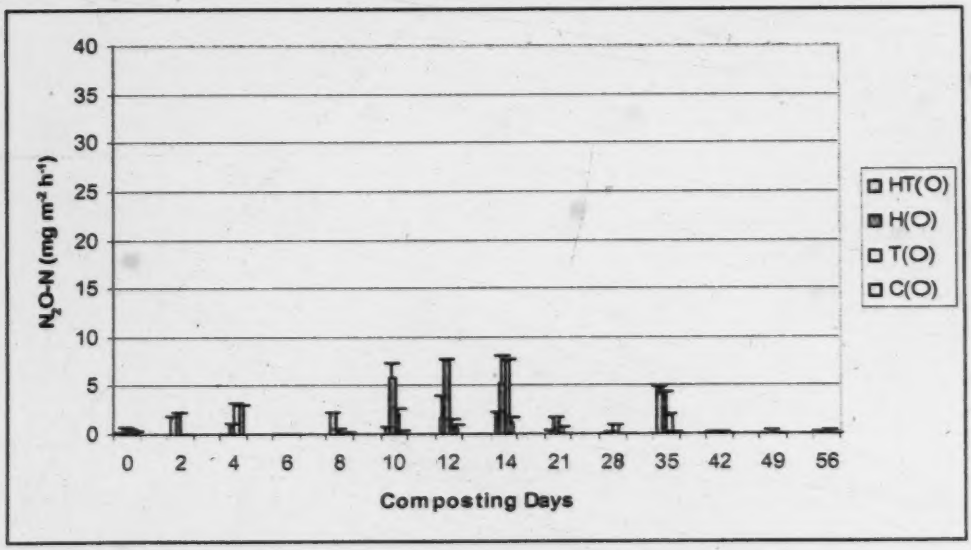

Figure 3.15: N_2O-N Emissions vs. Composting Days – Indoor Vessels Laboratory Trial

3.4 Discussion

3.4.1 Physical Properties

The results obtained from FAS and BD profile determinations (Figures 3.1 and 3.2) using air pycnometry are consistent with the results obtained by Agnew (2002). As depth increased, BD increased, while FAS decreased. Side-by-side comparison of Figures 3.1 and 3.2 reveals a reciprocating relationship between FAS and BD also noted by Agnew (2002). At greater depths, simulated by a greater compressive load upon the compost, air voids are eliminated, causing a decrease in FAS; the air voids are replaced by material (i.e. compost), thus causing an increase in BD. Included in Figures 3.1 and 3.2 are the moisture content values for each sample. The compost with the highest moisture content should be characterized as having the highest BD and lowest FAS. While treatment T (MC=51.6%) does display these characteristics the differentiation of the physical properties of the remaining three treatments are not readily apparent. This may be due to the heterogeneity of compost. The sample taken for moisture content analysis was only a very small portion of the total sample which was used in the physical property characterization. Therefore, the recorded moisture contents may not be representative of the entire samples.

The linear relationship observed in Figure 3.3 between FAS and BD also agrees with the findings of Agnew (2002). While only the HT treatment is presented in Figure 3.3, similar relationships were observed in the additional three treatments, with R^2 values ranging from 0.97 to 0.99.

The measurement of these physical properties of compost can be useful in explaining GHG emissions from composting. Monitoring these properties and adjusting them through various management strategies (e.g. turning, aeration, amendment with bulking agents) may lead to decreased emissions. For example, compost with high bulk density and low free air space has the potential to generate regions of anaerobicity, which are conducive to the production of CH_4 and N_2O (e.g. Czepiel (1996), He (2001)).

3.4.2 GHG emissions

The nature of the research performed makes it difficult to conduct replication of treatments. Time, space and budget constraints, as well as analytical equipment availability resulted in only one windrow of each treatment being established. The results obtained showed no significant differences ($P = 0.05$) between treatments.

Though a greenhouse gas, CO_2 generated by composting is not considered a net emission because of the assimilation of CO_2 by plants in the natural carbon cycle (Hellebrand, 1998). In other words, the amount of CO_2 emitted during composting of biomass is assumed to be offset by the amount of CO_2 consumed by this biomass for use in photosynthesis when it was living plant matter. Nevertheless, monitoring CO_2 emissions still serves as a qualitative indicator of compost quality, as it, along with water are the main biodegradation products during composting. Therefore, CO_2 emissions trends are often analogous to temperature trends; high during the thermophilic phases, and then gradually decreasing as the composting process gradually progresses through the mesophilic stage and finally, the curing stage.

Figures 3.4 and 3.5 present CO_2-C emission data from the four field piles, and Figures 3.10 and 3.11 present CO_2-C emission data from the laboratory trials. As in the first field season trial, CO_2-C emissions remained substantial throughout the 84-day composting trial. Variability in CO_2-C emissions (i.e. lack of a distinguishable trend) in the field trials was also evident, as in the first field trial. Also, the magnitude of CO_2-C emissions remained comparable with the first field season trial, with emissions generally ranging from 0-10 g CO_2-C m^{-2} h^{-1}. While cumulative emissions were of the same magnitude, the CO_2 emissions from the laboratory vessels exhibit more typical trends observed in other studies (Hao, 2001; Sommer and Moller, 2000), with higher emission rates near the beginning of the composting period, and less near the end of the trial. The field piles however, show more constant emissions of CO_2 throughout, with periodic spikes in emission rates, up until as late as the tenth week of composting (day 70).

However, in the laboratory trial, cumulative CO_2 emissions from the outdoor vessels were significantly lower ($P = 0.05$) than the cumulative CO_2 emission from the indoor vessels. While cumulative values ranged from 3.3 to 5.3 kg CO_2-C m^{-2} for the indoor vessels, CO_2-C values from the outdoor vessels ranged from only 0.9 to 1.6 kg CO_2-C m^2. This difference may have been due simply to the fact that the vessels outside were not insulated (in order to more accurately simulate outdoor windrows) and because of this, retained less of the heat generated from the composting process, resulting in lower rates of biodegradation and oxidative processes that would contribute to CO_2 emissions.

CH_4 is produced in compost when anaerobic conditions prevail. Figures 3.6 and 3.7 present the CH_4-C emissions data from the field trial, and Figures 3.12 and 3.13 present the CH_4-C emissions from the laboratory trials. Two of the main factors largely responsible for the facilitation of anaerobic environments are: 1) high oxygen uptake during the active phases of composting (Sommer, 2000) and; 2) increasing bulk density and loss of porosity due to settling of the compost, especially in static piles, where optimal porosity is a key factor in influencing the convective flow of oxygen into the pile. As in the case of CO_2, trends and cumulative values of CH_4 emissions are very comparable to those observed in the first field season (Chapter 2). Generally, the emissions measured from the field trials are an order of magnitude greater than those observed from the laboratory vessel. This may be explained by the fact that temperatures, and therefore biodegradation rates and oxygen consumption were greater in the field piles, thereby creating more oxygen deficient, methane-producing regions in the field windrows than in the laboratory vessels. Secondly, the physical size of the piles in relation to the laboratory vessels should be considered. The larger field piles are likely to compress as a result of their larger mass, reducing porosity and accelerating the formation of anaerobic regions within the compost. Also, it would have been easier for air to get into the laboratory vessels via the perforated floors than into the large field piles.

The CH_4 emission trends also differ greatly between the field and laboratory trials. In the field, CH_4 emissions increased through the first 2 weeks of composting as oxygen was consumed through the thermophilic phase of composting. The subsequent decrease and leveling of the CH_4 emission rates may indicate that the porosity of the piles was

sufficient to support, if not completely, at least a partially aerobic environment despite the lack of turning of the compost, as explained by Hellebrand (2001). Also, as the thermophilic phase came to an end, temperature and oxygen consumption dropped, making conditions less optimal for the propagation of methanogenic bacteria (Hellman, 1997). In the laboratory, the majority of CH_4 was emitted within the first 4 days of composting, and then decreased sharply. A possible explanation for this is that due to the small physical mass of the laboratory compost, an aerobic environment was able to be maintained. Also, a "drying out" of the surface layer of the laboratory compost was observed. This dry surface layer could act as a biofilter for any emissions generated within the compost and could result in the oxidation of CH_4 to CO_2 by methanotrophic bacteria (Sommer, 2000; Hellman, 1997).

N_2O is produced during nitrification and denitrification reactions (Czepiel, 1996). Current research suggests that the extent to which N_2O will "leak" from these reactions is largely dependent on the availability of inorganic nitrogen, organic carbon, and oxygen, and that N_2O is produced by both processes in compost under "moderate" oxygen concentrations (Barton and Atwater, 2002; He, 2001). Emissions from the field and laboratory are generally of the same magnitude. In both cases, the emission rates remained highly variable over the course of the trial as has been most commonly witnessed in previous research. Perhaps the seemingly random nature of N_2O emissions and the lack of distinguishable trends is a direct result of the number of factors influencing its production and emission. In the case of the laboratory trial, emission maxima were observed at 21 days, agreeing strongly with the observations of Hellebrand

(2001). However, numerical differences in N_2O emissions were evident within the laboratory trial. There was a notable increase in N_2O emissions when comparing the high C:N treatment to the low C:N treatment. This is contrary to the hypothesis that an increase in C:N would decrease N_2O emission due to the decrease in N available for nitrification/denitrification reactions. However, C:N ratio is not the only variable that needs to be considered regarding N_2O emissions. Examination of the physicochemical analysis of the compost treatments reveals that, while the "high C:N" treatments did have higher C:N ratios than the low C:N treatments, they also had higher NO_3^- concentrations (Table 3.4). NO_3^- availability is also a very important contributing factor to N_2O production, as it is the nitrogen species which undergoes denitrification. Thus, the higher NO_3^- concentrations could have offset the effects of high C:N ratios. It is also important to consider the type of carbon source added to the high C:N treatments which increased the C:N ratio. An addition of largely unavailable cellulose carbon in the form of wood waste was added to the compost. The fact that this carbon is unavailable likely means that it had little influence on N_2O production, as it is generally agreed upon that only by increasing available carbon (electron donor), will a decrease in N_2O produced by denitrification, be observed (Firestone, 1982). This decrease occurs because rapid electron donation from readily oxidizable C forces the denitrification reaction sequence to its terminal end product N2. Finally, there was a numerical difference in pH between the treatments, with the two high C:N treatments having slightly lower pH (Table 3.4). Another well established relationship concerning denitrification and N_2O production (Firestone, 1982) is that N_2O production from denitrification increases as pH decrease. It has been proposed that lower, more acidic pH levels may inhibit N_2O-reducing enzymes,

or that acidity may provide suitable conditions for the intermediates of the denitrification process to undergo reduction to N_2O. This relationship may also help explain why higher N_2O emissions were observed from the high C:N treatments.

3.4.3 Variability in Measurement of GHGs

The quantification of GHGs from composting is complicated by the potential for error in measurement, particularly when applying the static chamber collection/measurement technique to the field. Error bars within the presented data indicate the standard deviation from the mean of "paired" samples, and clearly shows an increase in error when taking field measurements. The more typical emission trends observed in the laboratory trials, coupled with relatively lower error in measurements indicate that closed chamber methods may be more suitable for laboratory applications than in the field.

There are two possibilities being explored in order to explain this difference: the external environmental influences of wind and sunlight. Wind blowing near a static chamber placed on compost may cause pressure fluctuations around or within the chamber, and cause fluctuations in the gases collecting in the chamber. Sunlight shining directly on the metal chambers may cause rapid heating of the chamber and its contents, causing an increase in pressure within the chamber, inhibiting natural flux rates.

3.5 Conclusions

While numerical differences were evident in emissions between treatments, they were small, and statistically significant differences were not observed. In general, emission

rates, and trends from the field and from the laboratory were comparable, suggesting that seasonal variation does not influence emission rates. Decreased variability in measurements from the laboratory versus field measurements indicates that static chamber technologies may be more suitable for controlled laboratory experiments than for field applications. A benefit of the controlled environment under which the laboratory experiments were conducted was the establishment of more well defined emission trends, particularly for CO_2. The comparability of CO_2 data from the field and laboratory may be a result of the limited amount of conditions required to be met to produce CO_2 from composting. Essentially, all that is required for CO_2 production is normal composting conditions, which can be more easily controlled. The difference in size of composting units was thought to partly explain the differences in CH_4 emissions, while the variability of N_2O is still poorly understood.

3.6 References

Agnew, J.M. 2002. The free air space and bulk density of compost. MSc thesis. Universtiy of Alberta, Edmonton, Alberta.

Ballestero, T.P. and E.M. Douglas. 1996. Comparison between the nitrogen fluxes from composting farm wastes and composting yard wastes. Transactions of the ASAE. 39:1709-1715.

Barton, P.K. and J.W. Atwater. 2002. Nitrous oxide emissions and the anthropogenic nitrogen in wastewater and solid waste. Journal of Enivironmental Engineering. 128:137-150.

Beck-Friis, B., M. Pell, U. Sonensson, H. Jonsson, and H. Kirchmann. 2000. Formation and emission of N2O and CH4 from compost heaps of organic household waste. Environmental Monitoring and Assessment. 62:317-331.

Beck-Friis, B., S. Smars, H. Jonsson, and H. Kirchmann. 2001. Gaseous emissions of carbon dioxide, ammonia and nitrous oxide from organic household waste in a compost reactor under different temperature regimes. J. Agric. Eng. Res. 78:423-430.

Czepiel, P., E. Douglas. R. Harriss. and P. Crill. 1996. Measurements of N_2O from composted organic wastes. Environ. Sci. Technol. 30:2519-2525.

FAO. 2001. Global estimates of gaseous emissions of NH_3, NO and N_2O from agricultural land. ISBN 92-5-104698-1. Available electronically at: www.fao.org/DOCREP/004/Y2780E/y2780e00.htm.

Firestone, M.K. 1982. Nitrogen in agricultural soils. F.J. Stevenson, Editor. American Society of Agronomy, Madison, WI.

Hao, X., C. Chang., F.J. Larney, and G.R. Travis. 2001. Greenhouse gas emissions during cattle feedlot manure composting. J. Environ. Qual. 30:376-386.

He, Y., Y. Inamori, M. Mizuochi, H. Kong, N. Iwami and T. Sun. 2001. Nitrous oxide emissions from aerated composting of organic waste. Environ. Sci. Technol. 35:2347-2351.

Hellebrand, H.J. 1998. Emissions of nitrous oxide and other traces gases during composting of grass and green waste. J. agric. Engng. Res. 69:365-375.

Hellebrand, H.J. and W.D. Kalk. 2001. Emission of methane, nitrous oxide, and ammonia from dung windrows. Nutrient Cycling in Agroecosystems. 60:83-87.

Hu, L., C.T. Youngberg and C.M. Gilmour. 1972. Readily oxidizable carbon: an index of decomposition and humification in forest litter. Soil Sci. Soc. Amer. Proc. 36:959-961.

IPCC. 2001. Climate change 2001: The scientific basis. United Nations Environment Programme website. Available electronically at: http://www.grida.no/climate/ipcc_tar/wg1/006.htm.

Jacinthe, P-A, W.A. Dick and L.C. Brown. 2000. Bioremediation of nitrate-contaminated shallow soils and water via water table management: evolution and release of nitrous oxide. Soil Biology and Biochemistry. 32:371-382.

Komilis, D.P. and R.K. Ham. 2000. A laboratory method to investigate gaseous emissions and solids decomposition during composting of municipal solid wastes. Compost Sci. Util. 8:254-265.

Laidlaw, J.W.A. 1993. Nitrous oxide emission during the soil thaw. MSc thesis. University of Alberta.

Lopez-Real, J. and M. Baptista. 1996. A preliminary comparative study of three manure composting systems and their influence on process parameters and methane emissions. Compost Sci. Util. 4:71-82.

McKeague, J. A. 1978. Manual on soil sampling and methods of analysis. Prepared by subcommittee on methods of analysis of Canadian Soil Survey Committee. 2nd edition. Canadian Society of Soil Science. Ottawa, Ontario, Canada

Sommer, S.G. and H.B. Moller. 2000. Emission of greenhouse gases during composting of deep litter from pig production – effect of straw content. Journal of Agricultural Science. 134: 327-335.

Sommer, S.G. and P. Dahl. 1999. Nutrient and carbon balance during the composting of deep litter. J. agric. Engng. Res. 74:145-153.

SAS. 2001. Version 8.02. Statistical Analysis System Institute Inc. Cary, NC, USA.

Technicon Industrial Systems. 1973. NH_4^+-N Industrial method #90-70W. Technicon autoanalyzer II procedures manual.

Technicon Industrial Systems. 1973. NO_3^--N Industrial method #100-70W-B. Technicon autoanalyzer II procedures manual.

UNFCCC. 1992. Kyoto Protocol to the United Nations Framework Convention on Climate Change. Available electronically at: www.ec.gc.ca/climate/kyoto.htm.

Zeman, C., D. Depken, and M. Rich. 2002. Research on how the composting process impacts greenhouse gas emissions and global warming. Compost Sci. Utilization 10:72.85.

CHAPTER 4

Evaluation of the Static Chamber Method for Collection of Greenhouse Gases from Composting

4.1 Introduction

The static chamber method is a collection method of off-gases from emitting surfaces. Using this method, a chamber of known volume and base area is placed on an emitting surface, and left to allow gases to accumulate within it. After a predetermined time, samples are drawn from the chamber, and analyzed. Collected samples yield concentration values of the desired gases, and knowing this value, along with physical properties of the gas, dimensions of the chamber, and the time which gases were allowed to accumulate in the chamber, an emission rate can be calculated, expressing the mass of gas emitted per unit area, per unit time. Though the static chamber method is considered an easy-to-use, inexpensive and practical method for the collection of off-gases from emission sources, disadvantages to this method have been identified (FAO, 2001). One such disadvantage is that the use of completely closed static chambers may result in over- or underestimation of emissions. Overestimation of emissions can result if gases concentrate within the closed chambers, and underestimation occurs when installation of a chamber causes alteration of the pressure fluctuations above the emitting surface, thereby inhibiting normal emission rates. Additionally, Sey (2003) points out that closed chambers, if not properly sealed on the emitting surface, may remain slightly open at the base of the chamber, where backward diffusion could take place. While the shortcomings of this method are often noted, static chambers continue to be used widely in the measurement of surface emissions from soils (e.g. Kabwe et al, 2002; Bajracharya et al., 2000). In fact, a study by Ambus et al. (1993) concluded that, for N_2O-N emission

measurements from arable soils, a closed chamber method was equally adequate a method for measuring emissions as much more expensive and complicated micrometeorological approaches, such as the use of tunable diode laser analyzers (TDLAs).

The static chamber method has also been used in the less-established field of greenhouse gas emission measurement from composting. Hao (2001) used a static chamber method where samples were drawn from chambers at 0, 5, 10, 20 and 30 minutes. From a plot of gas concentration versus time, a polynomial regression was applied to the data, and taking the derivative a $t = 0$ yielded an emission rate. Limited data exist in this field, and the deficiencies of the chamber method expressed by the soil science community have not been addressed. Of particular methodological importance is the length of time for which the chamber should be left on the compost before sampling. Optimization of this parameter greatly reduces the risk of over- and underestimation of emission rates. This issue has only been addressed briefly by Hellebrand and Kalk (2001), who assumed that the maximum allowable accumulation time for off-gases collecting in a closed chamber was 10 minutes. However, this was only an assumption, and not tested experimentally.

In addition to this concern, the suitability of the closed static chamber for use in the collection of greenhouse gases from composting remains another source of contention. Sampling off-gases from compost and from soil are different for a number of reasons. Firstly, compost is a much more biologically active material than soil, resulting in gaseous emissions orders of magnitude higher than those observed from soil. Secondly,

sampling of soil off-gases normally occurs on dense, flat surfaces, sampling of off-gases from composts is done on porous, irregular surfaces. The porous nature of compost induces channeling of emissions, and increases spatial variability of emissions, which may be difficult to account for when sampling with static chambers. Sampling on irregular surfaces with static chambers could result in gases diffusing out of the bottom of the chamber since spaces between the chambers and sampling surface may be present. Insertion of the chambers into the compost may reduce this effect, however, when conducting such measurements, minimal disturbance of the emitting surface is required.

As in the case of Hellebrand and Kalk (2001), a linear relationship between time and GHG concentration was assumed. Other non-linear gas emission models, such as the Hutchinson-Mosier model and the model proposed by Sey (2003) also are used to calculate emissions. While these models are useful in accounting for problems such as gas build-up and backward diffusion, assumptions made in these models and actions taken to limit problems such as backward diffusion are more easily applied to soil emissions than from composting. Therefore using a linear model provides a simplified method of calculating emission rates from compost. The objective of the research presented here was to determine the time range in which linear accumulations of GHG within the static chambers could be observed, in hopes of validating linear behaviour assumptions. Collection of the accumulation data also allowed for an assessment of the suitability of the static chamber method used for field and laboratory applications.

4.2 Materials and method

GHG emissions were measured from the same substrate in field piles and in 170 L laboratory vessels. Samples were taken from the turned treatments immediately after turning, from the research trial described in Chapter 3 (Teshima, 2003). The static chambers (chamber volume = 3.11 L, chamber base area = 0.0184 m^2) were placed on the compost, and samples were drawn at 0, 5, 10, 15, 20 and 30 minutes using 30 mL Teflon syringes fitted with 38 mm (1.5")-25.5 gauge needles. 20 mL of gas were then injected into each 10 mL evacuated Exetainer® sample vial (Labco Limited, High Wycombe, United Kingdom) and stored at 4°C until analysis.

Samples were analyzed for CO_2, CH_4, and N_2O using a gas chromatograph (Varian 3400, Varian Inc., Palo Alto, CA) The GC was equipped with a flame ionization detector (FID) for the detection of CO_2 and CH_4, and an electron capture detector (ECD) for the detection of N_2O. A Porapak N precolumn was used, and a HayeSep D (DVB 80/100)/Molesieve column was used as an analytical column. The oven and injector temperature was set at 400°C, and the column temperature was isothermal at 60°C.

4.3 Results

Sampling of the turned windrows was conducted after each turning event. Therefore, data are available from the six turning events from the field, and the four turning events from the laboratory. Analysis of the samples yielded concentration values, which were then plotted against the time at which they were drawn. The following figures present representative data from both the field and laboratory. Figures 4.1 and 4.2 present data

for CO_2 from the field and laboratory trials, respectively. Figures 4.3 and 4.4 present data for CH_4 from the field and laboratory trials, respectively. Figures 4.5 and 4.6 present data for N_2O from the field and laboratory trials, respectively. The values plotted for the field trials were from samples collected on July 17, 2002, and samples taken from the laboratory vessels were collected on July 18, 2002. Correlation coefficients were generated for the laboratory data using linear regression. The two series in each graph represents data from two separate chambers. Curves drawn through the graphs from the field emission data serve only to better illustrate the variability of the concentrations measured, and to emphasize the similarity in the variability of the series of data, and do not serve as an estimation of concentrations expected between the sampling times.

4.4 Discussion

The data presented in the figures above are representative of the trends observed in all the data gathered, both from the field and laboratory experiments. Data from the field showed high variability, with virtually no positive linear correlation between time and concentration. The only exception came from samples collected on August 28 and September 11, 2002, when good linear correlation was achieved with all three GHGs.

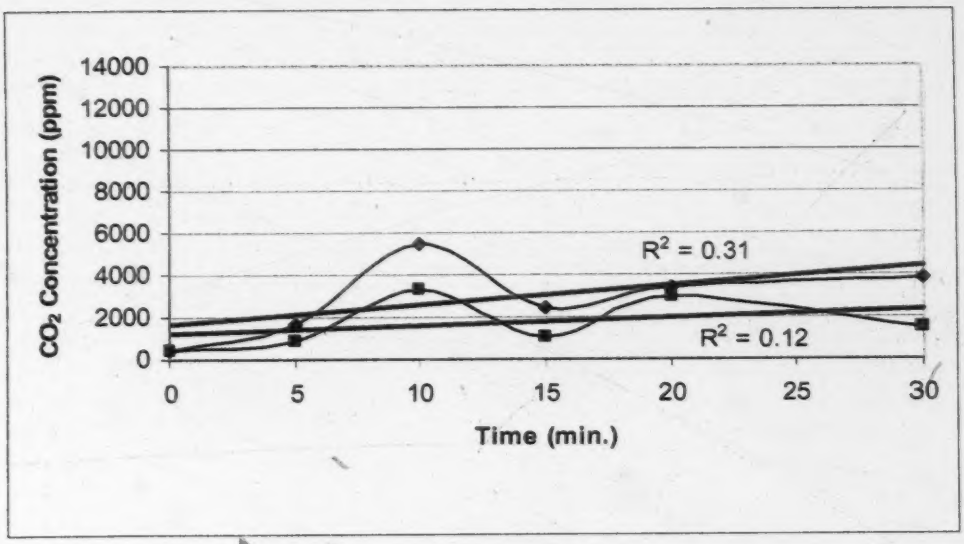

Figure 4.1: CO_2 (ppm) vs. Time (min.) – Field Trial

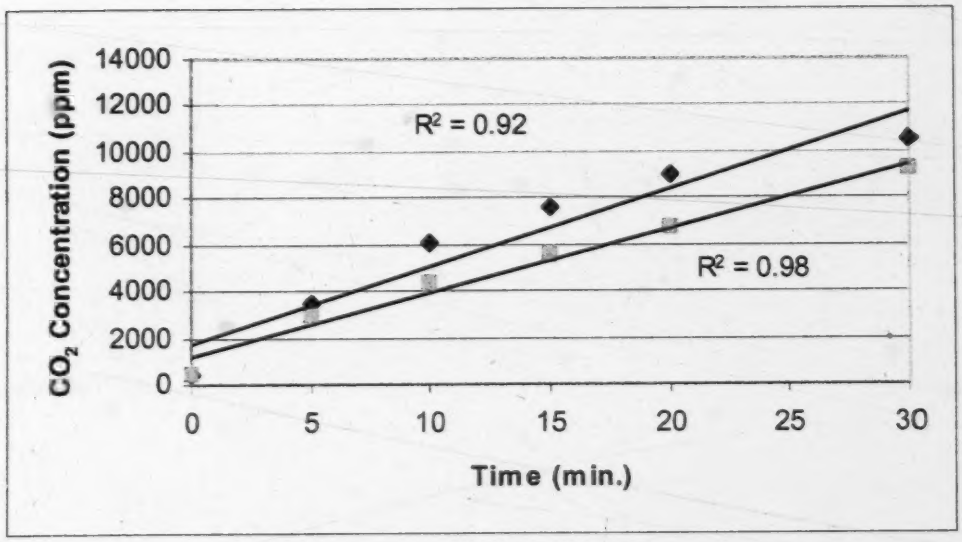

Figure 4.2: CO_2 (ppm) vs. Time (min.) – Laboratory Trial

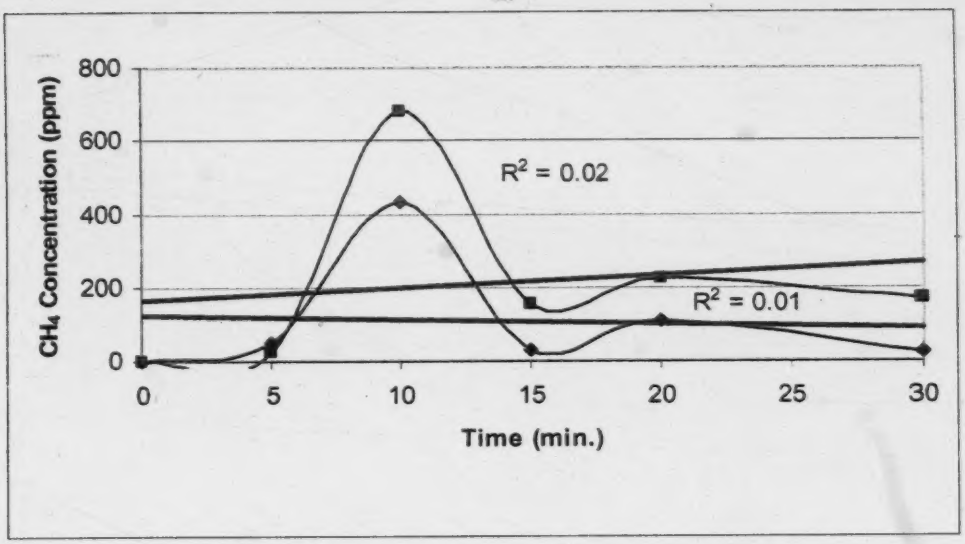

Figure 4.3: CH_4 (ppm) vs. Time (min.) – Field Trial

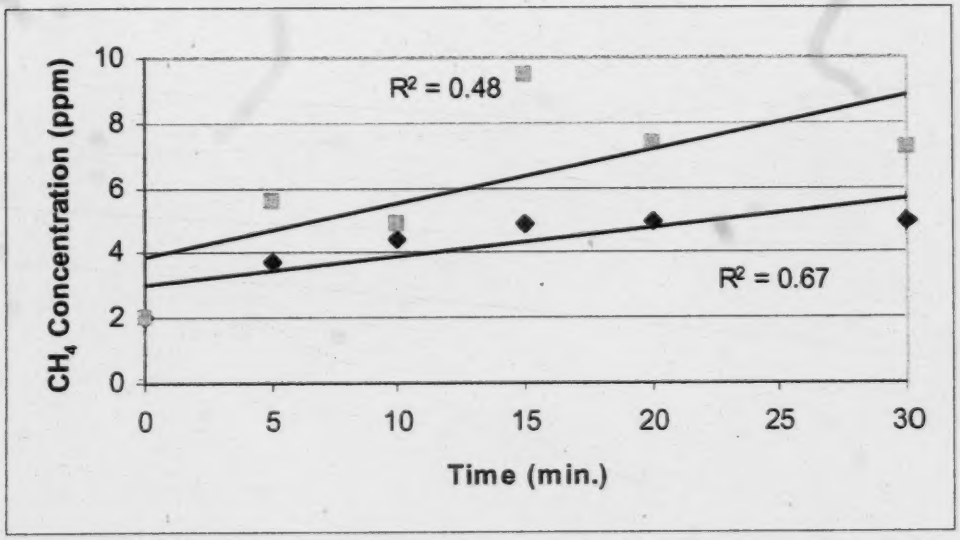

Figure 4.4: CH_4 (ppm) vs. Time (min.) – Laboratory Trial

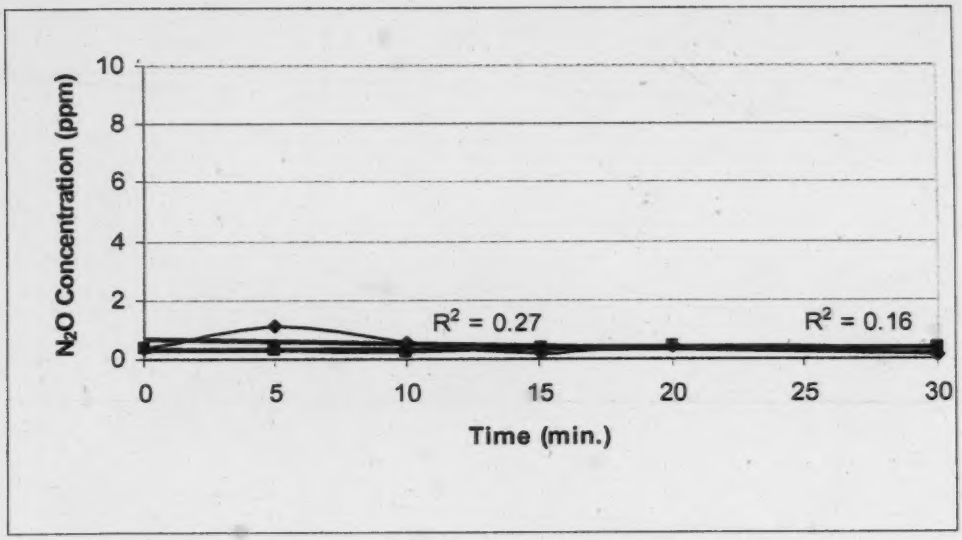

Figure 4.5: N_2O (ppm) vs. Time (min.) – Field Trial

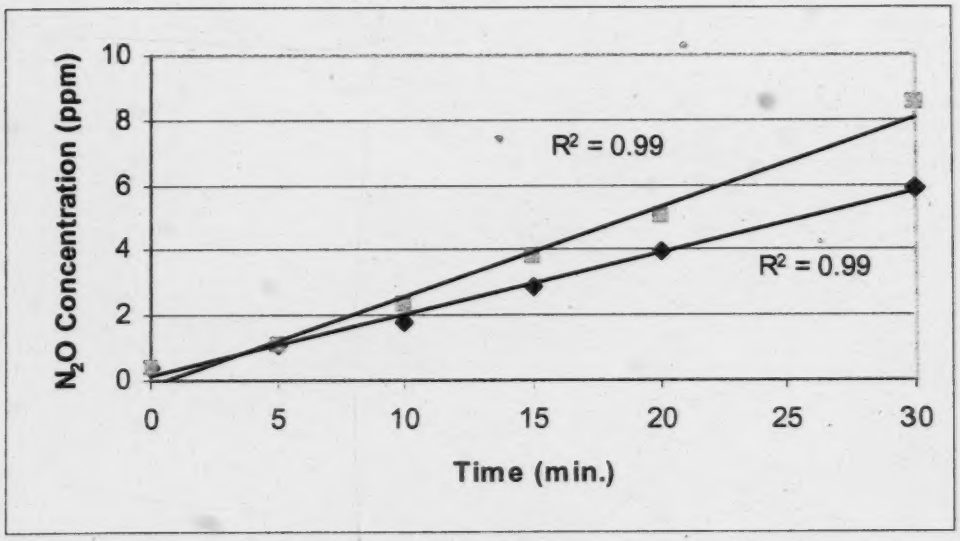

Figure 4.6: N_2O (ppm) vs. Time (min.) – Lab Trial

For example, R^2 values for the concentration vs. time graphs for CO_2, CH_4 and N_2O on September 11 were 0.97, 0.98 and 0.92 respectively.

The laboratory data consistently showed highly positive correlation between time and concentration, especially for CO_2 and N_2O, with R^2 values generally exceeding 0.95. CH_4, however, did not always show good linearity. The reason for the lack of linearity displayed for the CH_4 concentrations vs. time relationship is not readily apparent, however, it is possible that the sensitivity of the analytical methodology used in the detection of CH_4 may not have been adequate at the low levels measured for this experiment.

Initially, reasons for the difficulties encountered when trying to obtain linear data from the field were not clear, until it was noted how different conditions for sampling are in the field as opposed to the laboratory setting. In the laboratory, the vessels and chambers are housed in an environment that is kept essentially constant. Conversely, sampling conditions in the field are highly variable, and the chambers and compost windrows are exposed to environmental factors that could influence measurements.

The major environmental factor encountered in the field may be wind. Wind can cause pressure fluctuations around and within the chamber resulting in highly variable rates of accumulation of GHGs. Also, sunlight shining directly on the metal chambers may cause rapid heating of the chamber and its contents, causing an increase in pressure within the chamber, altering natural flux rates. These two factors impinging on the static chambers

may be responsible for the often oscillating relationship between GHG concentration and time observed in the majority of the field measurements.

Recent research conducted by Sey (2003) considered ways of improving closed chamber methods for the measurement of CO_2 from soils. Among the recommendation made were to insert the chamber into the emitting surface, in an attempt to "seal" the interface between the surface and the base of the chamber. Sey (2003) was able to demonstrate that flux measurement could be greatly influenced by the depth of insertion of the chamber, and recommended an insertion depth of 10 cm. Also, to counteract the influences of wind and the solar radiation on un-insulated chambers on the closed chamber, Sey (2003) recommended placing a box over the chamber to shield it.

While these appear to be practical solutions in emission measurements from soils, the degree to which they could improve chamber methods for emission collection in composting applications may be limited. The "seal" achieved by insertion of the chamber into the emitting surface would be more beneficial in soil, because it is a generally more compact, dense material. The porous nature of compost would make it difficult to achieve such a seal, and the spongy or elastic properties of compost would result in having to adjust chambers once inserted, causing additional site disturbance. Shielding chambers by placing a box over them may also prove more useful in soil applications than in compost. Shielding a chamber on a soil surface would essentially eliminate the influence of wind, because the soil surface is generally flat and not exposed, and soil is dense, minimizing the potential for airflow through the medium. Shielding

chambers on compost, on the other hand, is a much more difficult task. Even if the chamber were to be shielded using a box, the porous nature and typical location of a chamber on a windrow would still result in wind being a factor. Emission measurements from composting are generally taken from the top of a windrow, which, more than other parts of the pile, is exposed to environmental conditions such as wind. Although a box is covering the chamber, air will still be able to flow through the porous compost, possibly causing mixing of the contents within the chamber. Nevertheless, methods of shielding static chambers from environmental influences when working with compost needs to be investigated.

4.5 Conclusion

The data acquired from the laboratory suggest that in the cases of CO_2 and N_2O, linear approximation can be extended beyond the 10 minutes proposed by Hellebrand and Kalk (2001) to up to 30 minutes. When considering the other error associated with such measurements (e.g. spatial variability), the assumption of linear approximation (at least under controlled, laboratory conditions) provides a degree of simplification to measuring GHG emissions from a spatially, dimensionally and compositionally complex emitting source such as compost. Figures 4.1 to 4.6 provide graphic illustrations of the effects that external environmental factors may have on using the static chamber method in the field to sample GHGs from compost. While sampling in the laboratory was conducted under very controlled, stable conditions, the environment surrounding the field piles was highly variable, and appears to have introduced sources of error into measuring emissions from the field. When comparing field and laboratory data, it has become apparent that external

environmental conditions experienced in the field may lead to greater variability in measurement, and that when using the static chamber method in the field, great care should be taken to minimize the impacts of these conditions. Wind effects could be minimized by shielding the static chambers from the wind by enclosing the chambers in containers. The influence of sunlight on the chambers could be minimized by insulating the chambers, by constructing the chambers from materials that absorb less heat, or by shielding the chambers from the sunlight. Proposed recommendation for the improvement of emission measurements using closed chamber technologies may be more practical in soil applications than in composting.

Chamber techniques (Hao, 2001; Lopez-Real, 1996; Czepiel et al.1996 and; Hellman 1997) and similar methodologies have been used prior to the research presented in Chapters 2 and 3. However, the problems associated with such chamber use in the field apparently were not encountered or, at least, were not described by these authors. Though variability was clearly illustrated, particularly in Chapter 3, hourly and cumulative emissions were comparable to findings from literature whose methodologies were similar (Hao, 2001). It is therefore possible that, despite the fluctuations witnessed in field accumulation concentrations over time, an averaging or smoothening out of these deviations and subsequent generation of emission rates may still provide reasonable estimates of GHG emission from composting.

4.6 References

Ambus, P., H. Clayton, J.R.M. Arah, K.A. Smith, and S. Christensen. 1993. Similar N_2O fluxes from soil measured with different chamber techniques. Atmospheric Environment. 27A:121-123.

Bajracharya, R.M., R. Lal and J.M. Kimble. 2000. Diurnal and seasonal CO2-C flux from soil as related to erosion phases in central Ohio. Soil Science Society of America Journal. 64:286-293.

Czepiel, P., E. Douglas. R. Harriss. and P. Crill. 1996. Measurements of N_2O from composted organic wastes. Environ. Sci. Technol. 30:2519-2525.

FAO. 2001. Global estimates of gaseous emissions of NH_3, NO and N_2O from agricultural land. ISBN 92-5-104698-1. Available electronically at: www.fao.org/DOCREP/004/Y2780E/y2780e00.htm.

Hellebrand, H.J. and W.D. Kalk. 2001. Emission of methane, nitrous oxide, and ammonia from dung windrows. Nutrient Cycling in Agroecosystems. 60:83-87.

Hellmann, B., L. Zelles, A. Palojaervi and Q. Bai. 1997. Emission of climate relevant trace gases and succession of microbial communities during open-windrow composting. Applied and Environmental Microbiology. 3:1011-1018.

Kabwe, L.K., M.J. Hendry, G.W. Wilson, and J.R. Lawrence. 2002. Quantifying CO2 fluxes from soil surfaces to the atmosphere. Journal of Hydrology. 260:1-14.

Lopez-Real, J. and M. Baptista. 1996. A preliminary comparative study of three manure composting systems and their influence on process parameters and methane emissions. Compost Sci. Util. 4:71-82.

Sey, B.K. 2003. Problems associated with the closed chamber method of soil gas flux measurements. MSc thesis. Department of Renewable Resources, University of Alberta, Edmonton, AB.

CHAPTER 5

Synthesis

5.1 Summary

Demands for reduced anthropogenic greenhouse gas production (UNFCC, 1992) have resulted in the exploration for mitigative technologies in a number of sectors. In waste management, composting has been looked to as one such technology, as it has the potential to divert greenhouse gas-producing organic substrates from landfilling, while providing a beneficial, multi-purpose product. However, to justify the nomination of composting as a "greenhouse gas friendly" technology, the need for research, addressing emissions from composting is necessary. To date, the information available on this subject is extremely limited.

The research presented here attempted to contribute to the small pool of information on greenhouse gas emissions from composting by collecting baseline data. Familiarization with the methodology, the static chamber method, was carried out, and the spatial variability of greenhouse gas emissions from composting windrows was demonstrated. Secondly, emission measurements were conducted on windrows with varied physicochemical characteristics and management strategies, to observe whether altering these variables may influence greenhouse gas emissions. Concurrent with both field trials were laboratory trials, where it was examined whether emissions from small scale composting could be used as predictors of emissions from large-scale composting. Finally, an evaluation of the static chamber method as a suitable method for the collection of off-gases from composting was conducted.

5.2 Conclusions

The first field season (Chapter 2), served as a trial for familiarization with the static chamber methodology, as well as a probing experiment, quantifying GHG emissions from a single compost pile, established and maintained under typical management practices at a commercial composting facility. The static chamber technique used proved to be a very inexpensive and practical means of collecting off-gases from composting. As anticipated, the three primary GHGs, carbon dioxide, methane and nitrous oxide were all in evidence, and emitted at rates in agreement with the available literature (e.g. Hao, 2001). Emission measurements taken at various positions on the composting windrow indicated that the majority of emissions originated from the top, or peak of the compost, while emission rates from the sides, or slopes of the compost were essentially negligible. This was attributed to a "chimney effect" where the majority of emissions diffuse vertically through the top of the compost pile, while fresh air is drawn into the compost through the sides and the bottom of the compost. Comparison of the emission values and trends in the field to those observed in the laboratory yielded mixed results. While the magnitude and general emission trends between the field and laboratory were comparable for CO_2 and N_2O, emissions of CH_4 in the field were 1-2 orders of magnitude greater than those observed in the laboratory, and had opposing trends.

The second field season attempted to observe differences in GHG emissions from compost windrows with varied physicochemical properties and management strategies. In the field, no significant differences were observed between emissions from any of the treatments, for all three GHGs. There were, however, significant differences in average

emissions for all three of the GHGs between vessels kept inside and vessels kept outside in the laboratory trial. Results obtained from the field trial were comparable with those collected in the first field season, and were in agreement with a number of literature sources (e.g. Czepiel (1996); Hellmann (1997); Hellebrand and Kalk (2001); Hao (2001)). The effect of seasonal variation on GHG emission from composting appears to be negligible. While emissions differed for each gas numerically between treatments, the need for replication of such experiments is necessary in order to observe statically significant results.

Finally, further research on the static chamber method was conducted, in order to evaluate its suitability as a collection method for GHGs from composting. An experiment was performed to observe the time period in which linear accumulation of the GHGs could be collected within the chambers. While only one study had addressed this issue, claiming that linear approximations could be applied up to 10 minutes (Hellebrand and Kalk, 2001), the findings presented in Chapter 4 indicate that, under controlled laboratory conditions, this linear approximation range can be extended to up to 30 minutes, particularly for CO_2 and N_2O. Results from the field indicate that environmental factors such as wind and sunlight interfere with the effectiveness of the static chamber method, causing large fluctuations in accumulation trends. Unless practical solutions are found to counteract these effects, other collection techniques should be considered for field trials.

5.3 Future directions

The limited amount of data available on GHG emissions from composting allows for a number of research initiatives to be explored. Most importantly, continuation of the collection of baseline emission data will allow more accurate quantification of typical emission values and trends. Such information will be useful to the waste management sector for comparative purposes when attempting to promote composting as a "greenhouse gas friendly" technology. Concurrent with the collection of baseline data, the refinement and standardization of analytical and collection methodologies should also be addressed. Included within this standardization would be consistency in reporting results (e.g. compost physicochemical characteristics and emission units). This would increase the ease and validity of comparing results between experiments.

Specific to the research presented here, there is a need to employ replication in future trials. Although the majority of literature dealing with this subject does not present statistical analysis (most literature is focused simply on quantifying emissions), the probability of finding statistically significant results is improved by replicating experiments. Modification of the chamber method to make it more suitable for field applications is also an area which may be explored. The use of micrometeorological analytical methods, while costly, also remains an option.

5.4 References

Czepiel, P., E. Douglas. R. Harriss. and P. Crill. 1996. Measurements of N_2O from composted organic wastes. Environ. Sci. Technol. 30:2519-2525.

Hao, X., C. Chang., F.J. Larney, and G.R. Travis. 2001. Greenhouse gas emissions during cattle feedlot manure composting. J. Environ. Qual. 30:376-386.

Hellebrand, H.J. and W.D. Kalk. 2001. Emission of methane, nitrous oxide, and ammonia from dung windrows. Nutrient Cycling in Agroecosystems. 60:83-87.

Hellmann, B., L. Zelles, A. Palojaervi and Q. Bai. 1997. Emission of climate relevant trace gases and succession of microbial communities during open-windrow composting. Applied and Environmental Microbiology. 3:1011-1018.

UNFCCC. 1992. Kyoto Protocol to the United Nations Framework Convention on Climate Change. Available electronically at: www.ec.gc.ca/climate/kyoto.htm.